

**REMEDIAL INVESTIGATION (RI) REPORT**

**for**

**LOWER DARBY CREEK AREA SITE, CLEARVIEW LANDFILL  
OPERABLE UNIT 1 (OU-1)**

**Delaware and Philadelphia Counties, Pennsylvania**

**EPA Contract Number EP-S3-07-04  
EPA Work Assignment Number 023-RICO-D366  
Tetra Tech Project Number 01067**

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**MAY 2010**

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## **EXECUTIVE SUMMARY**

### **INTRODUCTION**

The U.S. Environmental Protection Agency (EPA) tasked Tetra Tech to perform a Remedial Investigation (RI) at the Lower Darby Creek Area (LDCA) site located in Philadelphia and Delaware Counties, Pennsylvania. The LDCA site was placed on the final National Priorities List (NPL) in June 2001 due to its potential release of hazardous substances to the nearby surface water, posing a threat to human health, ecological receptors, and other sensitive environments such as John Heinz National Wildlife Refuge (NWR) at Tinicum, which is the largest remaining freshwater tidal wetland in Pennsylvania.

The LDCA site consists of two separate landfill areas and is divided into two operable units by EPA as follows:

- Operable Unit 1 (OU-1): Clearview Landfill
- Operable Unit 2 (OU-2): Folcroft Landfill and Annex

EPA is the lead agency conducting RI activities for the Clearview Landfill site (OU-1), and a group of potential responsible parties (PRPs) are responsible for performing RI activities for the Folcroft Landfill and Annex site (OU-2). Accordingly, this RI report was prepared to present information pertaining to the Clearview Landfill site only. The general objectives of this RI were to characterize site conditions, determine nature and extent of contamination, and assess risks to human health and the environment. Findings from the RI serve as a basis to develop, screen, and evaluate remedial alternatives to address any unacceptable risks posed by OU-1.

Clearview Landfill is located along the eastern bank of Darby and Cobbs Creeks, at 83<sup>rd</sup> Street and Buist Avenue. The historical landfill footprint currently resides partly in Delaware County and partly in Philadelphia County, and includes the Clearview Landfill and the City Park east of the landfill.

Clearview Landfill was privately owned and operated without a permit from the 1950s to the 1970s by the Clearview Land Development Corporation, and used for the disposal of municipal and industrial waste collected from the City of Philadelphia and portions of Delaware County. In August 1973, due to several violations of state regulations related to land disposal and the absence of a landfill permit, the Pennsylvania Department of Environmental Protection (PADEP) took court action against the Clearview Land Development Corporation, and ordered it to cease all waste disposal activities at the landfill and follow a prescribed closure plan. However, even after this order, the property continued to be used for other waste disposal operations for many years.

Historical aerial photographs showed that when Clearview Landfill was closed in 1973, the landfill had expanded to the east and covered approximately 65 acres. The wetland areas formerly located east of the landfill were filled. Pools of standing liquid and pits containing liquid (the constituents of the liquid were not determined) were observed on the landfill surface. Tank cars (tanks) and dark stains were also noted on the landfill, indicating that liquid wastes may have been brought to the landfill. The aerial photographs also showed that new residential properties were constructed east and southeast of the landfill, possibly on top of a formerly filled area.

Currently, the southern end of the landfill is used by several businesses, including a waste transfer station, a truck/equipment storing and snow plowing business, an auto repair and salvage operation, and a drum recycling operation. Local residents access the landfill area for walking, all terrain vehicle riding, deer hunting, and other activities. Abandoned cars have also been found at the landfill.

## **SITE CHARACTERISTICS**

The LDCA site is located in an industrial section of Darby Township, Delaware and Philadelphia Counties in Pennsylvania. The year 2000 census data show that there are about 2,300 people living immediately adjacent to Clearview Landfill and 755 homes in the 6-block area between S. 84<sup>th</sup> and S. 78<sup>th</sup> Streets of Linbergh Blvd. All residential properties are located within the City of Philadelphia boundary.

In general, land use near the LDCA site is urban residential mixed with commercial, industrial, and natural area uses. According to the 1983 study of the Clearview Landfill by EPA, the land uses of the Clearview Landfill are known as Commercial/Light Industrial; Vacant Urban Lands; and Dump, although the entire land has been used for dumping waste. Residential land use is predominant in close proximity to the east of Clearview Landfill.

OU-1 includes three sub-areas: the Clearview Landfill, the City Park located adjacent to the landfill, and the Eastwick residential community. Clearview Landfill has the largest relief of any nearby land surface, rising to an elevation of over 80 feet above mean sea level (msl), and is currently situated within Delaware County. However, the historical footprint of the landfill operation was known to extend to the City Park area, which is deeded in Philadelphia County (or City of Philadelphia). The adjacent City Park is currently a public recreational facility, which includes tennis courts, basketball courts, playgrounds, and walking paths.

All known residents in the Delaware and Philadelphia Counties are supplied with potable water by a public water supplier. No drinking water wells are known to exist on the Pennsylvania side of the Delaware River. On the New Jersey side of the Delaware River, drinking water wells are known to exist for Gibbstown and the Borough of Paulsboro (located 5.5 and 4 miles south of Clearview Landfill, respectively).

Surface water features associated with the LDCA site consist of streams and marsh areas. Streams in the area include Darby, Cobbs, and Hermesprota Creeks. The main stem of Darby Creek originates in Easttown Township, Chester County and is joined by a number of tributaries as it flows downstream. Cobbs Creek, the major tributary of Darby Creek, converges with Darby Creek north of Clearview Landfill. Darby Creek is then joined by Hermesprota Creek near marsh area in John Heinz NWR at Tinicum. Water from Darby Creek and the marsh ultimately flows into the Delaware River. The confluence of Darby Creek and the Delaware River is approximately 3.5 miles downstream of Clearview Landfill. An impoundment and tidal wetlands exist within the John Heinz NWR.

Tidal influence exists throughout the lower portion of Darby Creek and upstream as far as Clearview Landfill. On average, Darby Creek is tidal up to the confluence of Darby Creek and Cobb Creek, located near the northern portion of the landfill, but the extent of tidal influence changes depending on climate conditions.

Flood plains encroach significantly onto the study area. Hurricane Floyd in 1999 caused significant flooding throughout the Cobbs and Darby Creeks, including the Eastwick neighborhood and surrounding area, inundating many homes. It appears that flooding is a common event in the area.

Although the LDCA site is located within a highly urbanized industrial setting that could negatively affect habitat quality in the area, there are several significant and unique habitat areas adjacent to the site. The John Heinz NWR (the Refuge) at Tinicum consists of approximately 1,200 acres of wetlands within two miles of Darby Creek near the LDCA site and represents the largest freshwater tidal marsh remaining in Pennsylvania. The Refuge is home to a variety of wildlife. Birdwatchers have recorded more than 280 species of birds in and around the Refuge. Clearview Landfill is located on a major waterfowl migration route that is part of the

Atlantic Flyway. The 145-acre shallow impoundment in the John Heinz NWR is connected to part of the tidal marsh via a tidal gate. This large open water along with the adjacent heavily vegetated tidal wetland forms an ideal habitat for migratory waterfowl. In addition, the Refuge is one of the few places in Pennsylvania where the state-endangered red-bellied turtle and southern leopard frog can be found. Moreover, Cobbs and Darby Creeks are listed as warm-water fishing streams by the Pennsylvania Fish and Boat Commission (PFBC).

The habitat quality within/near Clearview Landfill is considered poor because of constant disturbance, poor natural food sources (except for scavengers), and lack of quality cover on made-land soils, all resulting from urbanization. However, the on-site wooded area along with the riparian corridor does represent habitat that is unique to the urbanized setting.

## **SITE GEOLOGY AND HYDROGEOLOGY**

Clearview Landfill is situated on unconsolidated coastal plain sediments overlying bedrock of the Wissahickon Formation. Bedrock was encountered at depths ranging from approximately 18 feet below ground surface (bgs) in the Eastwick neighborhood to as deep as approximately 44 feet bgs in the southern industrial area of the landfill.

From the surface layer downward, Clearview Landfill has fill soil, concrete, and construction debris up to approximately 20 feet thick at ground surface; landfill wastes up to 75 feet thick at the center of the landfill; a discontinuous peat layer (an organic-rich marsh deposit) with a thickness of 1-3 feet; 10-15 feet thick layers of sand, gravel, silt, and clays (similar in description to natural Trenton Gravel deposits); and Wissahickon Formation rock, consisting of micaceous schist.

Geology in City Park has 1-2 feet thick fill soil at ground surface (in some areas, particularly the northern open field of City Park, there was only a very thin soil fill cover and wastes visibly protrude at ground surface.); landfill wastes with 8-12 feet thickness in the former marshland below the City Park area; a discontinuous natural organic peat layer (an organic-rich marsh deposit) with a thickness of 1-3 feet; discontinuous sand, silts, and clays, with a total thickness of 10-15 feet above bedrock; and Wissahickon Formation rock, consisting of micaceous schist.

The Eastwick Neighborhood has re-worked fill soil and demolition debris in 1-2 feet thickness (thicker in some isolated places) at ground surface (demolition debris appears to be a relic of demolition of structures that pre-dated the current Eastwick townhouse construction in the mid-1970s); a 15-25 feet thickness (in total) layer of discontinuous sand, silts, and clays above bedrock; and Wissahickon Formation rock, consisting of micaceous schist.

A groundwater mound (or high water table) exists under the landfill and produces radial groundwater flow away from Clearview Landfill. Regionally, groundwater flow is expected to be southwest toward the Delaware River. However, locally near Clearview Landfill, groundwater flows radially outward from the landfill toward the Darby and Cobbs Creeks, south below the southern industrial area, and east below the Eastwick neighborhood.

Groundwater recharge occurs throughout the Clearview Landfill area. Visually, little or no runoff occurs from Clearview Landfill during smaller storm events. Based on water level and groundwater flow directions, groundwater recharge occurs primarily in enclosed drainage basins (on the east side of the landfill) that do not drain into Darby and Cobbs Creeks.

Groundwater and/or leachate visibly discharges at seeps in the banks of the Darby and Cobbs Creeks north, west, and southwest of the landfill. Gas bubbles were observed during the RI in the base of the Darby and Cobbs Creeks, suggesting that groundwater/leachate seeps into the creek bed with actively

decaying organic matter.

The City Park and part of Eastwick neighborhood were originally a wetlands/marsh area. Hydrologically, Clearview Landfill was a regional groundwater discharge area and groundwater flows regionally southeastward, but locally toward Darby and Cobbs Creeks.

Water table elevations in shallow wells appeared to be higher than those in deeper wells. Therefore, it is likely that hydraulically separate zones exist above/below discontinuous silt/clay layers. Groundwater below these layers is semi-confined and is not directly connected to the water-table aquifer in these areas. However, north and east of Clearview Landfill, fewer silt/clay confining layers exist below the water table, and semi-confined conditions may not be present.

## **RI FIELD ACTIVITIES**

During the RI, field activities were conducted in several distinct areas of interest associated with OU-1: three investigative zones, including Clearview Landfill (Zone 1) and City Park (Zone 2) within the historical landfill footprint, and adjacent residential area, Eastwick neighborhood (Zone 3); Darby and Cobbs Creeks; and John Heinz NWR. The nature and extent of contamination were determined in each area. In addition, the actual or potential risks resulting from the presence of contamination in each medium of concern were estimated to provide information for determining appropriate cleanup measures, if warranted.

Field activities during the RI included the following major elements in chronological order:

- Sediment and surface water sampling in Cobbs and Darby Creeks, upstream and downstream of Clearview Landfill (Spring/Summer 2002).
- Leachate seep sampling from the bank of Clearview Landfill along Darby Creek (Summer 2002).
- Installation of soil borings in City Park and Eastwick neighborhood, and monitoring wells in City Park, followed by soil, soil/landfill gas, and groundwater sample collection (first round - Winter/Spring 2003).
- Installation of staff gages along Darby and Cobbs Creeks for the tidal study (January 2003).
- Sampling of stormwater and ponded water in City Park (Spring 2003).
- Installation of additional soil borings in City Park and Eastwick neighborhood, and permanent vapor monitoring wells in City Park, followed by soil, soil/landfill gas, and groundwater sample collection (second round - Summer 2004).
- Collection of earthworm, and sediment and surface water sampling performed in City Park, and Darby and Cobbs Creeks for ecological assessment (Fall 2005).
- Installation of soil borings and groundwater monitoring wells in Clearview Landfill (Winter 2005/2006). Samples of soil, soil/landfill gas, and groundwater were collected.
- Landfill leachate seep sampling from the banks of Clearview Landfill along Darby Creek (Spring 2006).

- Groundwater sampling from all permanent monitoring wells (second round – Summer 2006).

## **NATURE AND EXTENT OF CONTAMINATION**

### *Soil and Landfill Gas Assessment*

Analytical results from multiple sampling events in 2003 through 2006 indicate that contaminants, primarily volatile organic compounds (VOCs) and landfill-related gas components were present in soil gas throughout the Clearview Landfill area.

Methane, a principal landfill gas, was detected widely in the soil gas sampling locations on/near Clearview Landfill, but consistently at much lower concentrations than would be expected from an active or recently closed municipal solid waste (MSW) landfill (typical of 40 to 60%). Eleven soil gas sampling locations were detected with comparatively high methane content (equal to or greater than 5% by volume), ranging from 5% to 75.5% by volume. Of these, methane content at nine locations was within the flammable range (between 5 to 15% by volume).

Soil gas was detected with many fuel- and solvent-related substances at concentrations exceeding their respective EPA Region 3 Risk-Based Concentrations (RBCs) for ambient air. Of these, benzene was the most frequently detected substance in soil gas throughout the Clearview Landfill and the surrounding areas. Within Clearview Landfill, comparatively high levels of VOCs were detected in the central and southern parts of the landfill. More significant levels of VOCs were detected in the northern open field and central parts of City Park. Levels of VOCs detected in Eastwick neighborhood area were not significant.

### *Stream and Leachate Sediment Assessment*

Of the contaminants detected in the sediment samples from Darby and Cobbs Creeks, polycyclic aromatic hydrocarbons (PAHs) were detected more frequently and at much higher concentrations than other contaminants. Concentrations of these PAHs exceeded their respective screening values (i.e., the EPA Region 3 RBCs for residential soil that were modified for sediment or the Biological Technical Assistance Group [BTAG] Screening Benchmarks, a.k.a., ecological screening value [ESV]).

VOCs and pesticides were detected in relatively low concentrations in stream sediment samples. Arsenic was detected above its screening value at stream sediment sampling locations downstream of the landfill. Many other metals were also detected above their respective screening values in sediment samples from Darby Creek near Clearview Landfill.

Leachate sediment samples collected at leachate seep locations along Darby and Cobbs Creeks indicated that the seep sediments had elevated concentrations of metals and PAHs, and relatively low concentrations of other semivolatile organic compounds (SVOCs), pesticides, and VOCs. Many PAHs in the leachate sediment samples were detected over their respective screening values. The leachate sediment samples also contained elevated levels of metals, especially aluminum and iron. However, aluminum concentrations were below its screening values (78,000 mg/kg). Arsenic was detected in exceedance of the screening value (4.3 mg/kg of RBC or 9.8 mg/kg of ESV) in five of the ten sediment samples collected at the seep locations.

Sediment samples from the John Heinz NWR had high concentrations of metals (e.g., arsenic) and PAHs, and relatively low concentrations of pesticides and VOCs. The sediments in Darby and Cobbs Creeks upstream of Clearview Landfill had elevated concentrations of metals and PAHs, and relatively low concentrations of pesticides and VOCs. In addition, the total concentration of PAHs in sediment was

higher in Cobbs Creek upstream of the landfill than in Darby Creek upstream of the landfill. The widespread elevated background (upstream) concentrations of PAHs, other organic compounds, and metals were expected, given that the watershed is highly urbanized and receives discharges from industrial and municipal processes, combined sewer overflows, sanitary sewer overflows and stormwater and urban non-point drainage.

#### *Surface Water and Aqueous Leachate Assessment*

In general, the surface water samples contained far fewer analytes at much lower concentrations than the sediment samples. VOCs and SVOCs were not detected at elevated concentrations. Low levels of PAHs were detected above or equal to their screening values at four locations in Cobbs Creek upstream of the landfill, at one location in Darby Creek upstream of the landfill, and at one location in Darby Creek downstream of the landfill. This indicates that PAHs are mostly partitioned in sediment, as discussed previously due to their nature of strong affinity to sediment. Surface water in Darby Creek upstream of the landfill did not exhibit any VOCs and SVOCs (other than PAHs) above screening values.

Many inorganics in surface water samples were detected above their respective RBCs or ESVs. All sampling locations with arsenic detection exhibited total and dissolved arsenic concentrations greater than its RBC (0.45 µg/L). Total arsenic concentrations above its ESV (5 µg/L) were detected at four locations within the impoundment at John Heinz NWR and one location in Cobbs Creek upstream of the landfill. However, no samples were detected with dissolved arsenic above its ESV.

Surface water samples taken from two upstream locations in Cobbs Creek contained 5-day biochemical oxygen demand (BOD<sub>5</sub>) greater than the State Water Quality Standard of 10 mg/L. Surface water in Darby Creek, both upstream and downstream of Clearview Landfill, was relatively low in BOD<sub>5</sub>. However, four downstream sample locations within the impoundment at John Heinz NWR contained BOD<sub>5</sub> greater than the State Water Quality Standard.

Three upstream water samples in Darby Creek revealed fecal coliform concentrations greater than the State Water Quality Standard and the Delaware River Basin Commission regulation (200/100 mL). The furthest upstream location of the Clearview Landfill in Cobbs Creek exhibited the highest fecal coliform concentration (6,400/100 mL). Fecal coliform was also significant in seven out of nine downstream surface water samples.

The aqueous leachate samples collected from the banks of Clearview Landfill along Darby and Cobbs Creeks did not contain many organic compounds such as SVOCs, PAHs, VOCs, and pesticides. However, these samples contained elevated concentrations (above their respective screening values) of numerous inorganics including: aluminum, arsenic, barium, beryllium, cadmium, copper, iron, manganese, mercury, nickel, selenium, silver, thallium, vanadium, and zinc. Therefore, many metals found in the surface water appeared to be leachate-related.

The leachate BOD<sub>5</sub> results ranged from 32.1 to 378 mg/L, while chemical oxygen demand (COD) ranged from 75 to 1,800 mg/L. These results were fairly low for landfill leachate, indicating that the seeps may consist of large amounts of groundwater (or infiltration) mixed with the leachate.

#### *Soil Assessment*

Arsenic was encountered above its RBC for residential soil (0.43 µg/L) in all surface soil samples, but below its ESV (328 mg/kg). The highest arsenic concentration was 42.2 mg/kg at a sampling location in the southern industrial area where several businesses are currently being operated (Zone 1). The second highest concentration of arsenic (20.3 mg/kg) was detected in the northern part of the Eastwick

neighborhood (Zone 3) outside the landfill. Elevated arsenic (15.2 mg/kg) was also detected in one location in the northern part of City Park (Zone 2).

Elevated lead concentrations in surface soil samples were detected in the northern part of City Park (Zone 2) and the southern industrial area within Clearview Landfill. Elevated PAHs were detected in surface soils within the landfill footprint, but were also scattered outside the landfill footprint. Elevated PAH concentrations were also clustered in the northern part of the City Park and in the southern industrial area where oily wastes were encountered during drilling.

Polychlorinated biphenyls (PCBs) were scattered among the surface soil samples, mostly inside the landfill footprint. Significant PCB concentrations were found in the southern industrial area near where leaky transformers were found. In addition, elevated PCB concentrations were clustered in the northern open field in the City Park area.

Current operations in the southern industrial area are producing visible spills/leaks observed during the RI. Elevated PAH concentrations were reported for the following areas: (a) truck repair area - two shallow soil samples were above Resource Recovery and Conservation Act (RCRA) characteristic hazardous waste concentrations (i.e. greater than 50,000 µg/kg of PCBs); (b) area with waste oil staining on the ground surface where free-product oil was found in boreholes; and (c) drum recycling area where spilled/leaked substances were observed on the ground.

Pesticides were wide-spread throughout the area, but they were detected more frequently and at higher concentrations in Clearview Landfill than City Park or Eastwick neighborhood areas. Surface soils in five locations within the southern industrial area of the landfill were collected and analyzed for dioxins because of the former incinerator operation in this area. Total dioxin concentrations in surface soil samples (ranging from 0.32 to 1.20 pg/g) exceeded neither of the USEPA Region 3 RBC (4.3 pg/g) and ESV (10,000 pg/g), nor the Removal Action levels of 430 pg/g. However, total dioxin concentrations in two subsurface soils (10-feet deep) exhibited 8.87 and 47.26 pg/g, exceeding the USEPA Region 3 RBC for residential soil of 4.3 pg/g.

Metals, SVOCs (PAHs), and pesticide/PCBs were the principal contaminants detected in subsurface soils at depths up to 32 feet in Zones 1 and 2.

#### *Non-Aqueous Phase Liquids (NAPLs) Assessment*

Water and soil samples collected from the boreholes in City Park had signs of floating NAPL and free-product. These borings were located approximately in the historic stream channel (wetlands), which existed prior to the commencement of landfilling activities. Oily soil and water were observed at borings located in the northern part of the City Park area, approximately where pits and junked autos were observed on historical aerial photography. Surface and subsurface soils in this area were found to be highly contaminated.

Free-product oil was found at a monitoring well located in the central portion of Clearview Landfill and at soil borings near the western edge of the landfill. The actual source of the free-product was not determined during the RI.

Soil borings and groundwater monitoring wells located within the active southern industrial area contained evidence of free-product oil. Free-product oil in this area is likely the result of previous dumping, current operations, or both. Based on the direction of groundwater flow, free-product oil detected in this area likely flows toward Darby Creek.



## *Groundwater Assessment*

Based on the analytical results of the groundwater samples from monitoring wells and borings, groundwater in the study area has been impacted by organic and inorganic contaminants originating from wastes in Clearview Landfill. Contaminated groundwater exists in all three zones (Zones 1 through 3).

Arsenic appears to be pervasive in shallow groundwater which extends into the Eastwick neighborhood outside the historical footprint of the landfill. Relatively high concentrations of benzo(a)pyrene in groundwater (greater than RBC for tap water) were generally limited to the southern industrial area in Clearview Landfill. However, groundwater in the northern part of City Park and the western bank of the landfill also contained elevated benzo(a)pyrene levels. High concentrations of benzene and chlorobenzene were detected in groundwater in Clearview Landfill and City Park.

In general, the groundwater contained landfill-related pollutants (e.g., metals, VOCs, SVOCs, pesticides, and PCBs) in low to moderate concentrations that exceeded the RBCs. Groundwater samples collected from Clearview Landfill (Zone 1) were detected with these contaminants at higher concentrations than those from City Park or Eastwick neighborhood (Zones 2 and 3). Groundwater below the southern industrial area was heavily contaminated.

## **HUMAN HEALTH RISK ASSESSMENT (HHRA)**

Risks were assessed separately for distinct areas of interest, including soil, soil gas, and groundwater in Clearview Landfill (Zone 1), City Park (Zone 2), and Eastwick neighborhood (Zone 3); surface water and sediment in Darby Creek, Cobbs Creek, Tinicum Marsh; and on-site leachate seeps. Potential human exposure routes are specific to these media and human activity patterns associated with each area of interest.

EPA has defined acceptable risks for carcinogens as within the range of  $10^{-6}$  and  $10^{-4}$  excess lifetime cancer risk and for noncarcinogens as a hazard index (HI) of less than 1.0 (HI is the sum of hazard quotients [HQs] for substances that affect the same target organ or organ system.) There are various uncertainties associated with any risk assessment. Most uncertainties identified for the HHRA will result in the potential for overestimation of risk for both the Reasonable Maximum Exposure (RME) and Central Tendency Exposure (CTE) scenarios. These uncertainties should be considered as part of any risk management decisions about the site.

The following is a summary of the results for each exposure scenario quantitatively addressed in HHRA, which summarizes risks of potential receptors exposed to media of concern within the area of interest and lists contaminants of concern (COCs) contributing to the risks.

### *Surface Soil Risks*

#### Clearview Landfill

Exposure to Zone 1 (landfill area) surface soil was associated with estimated cumulative cancer risks that exceeded the acceptable risk range for lifetime residents ( $6.7E-4$ ) and lifetime recreational persons ( $1.3E-4$ ). COCs that contributed to these cancer risks included arsenic (which was similar to background), PCBs, dieldrin, heptachlor epoxide, benz(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, dibenz(a,h)anthracene, and indeno(1,2,3-cd)pyrene.

For the child resident, adult resident, and recreational child, the maximum of the estimated target organ HIs exceeded 1.0. For the residential child, antimony was the largest contributor to unacceptable

noncancer HIs, with additional contributions from heptachlor epoxide, dieldrin, endrin, Aroclor-1254, cobalt, iron, thallium, and copper (the latter 3 metals noted at levels similar to background).

For the evaluation of lead risks, a predicted blood lead level above 10 µg/dL in less than 5 percent of the receptor population is considered protective. Blood lead concentrations were predicted to exceed 10 µg/dL in more than 5 percent of an exposed population of child residents, but predicted blood lead levels were acceptable for other receptors. Soil lead concentrations exceeded background.

#### City Park

Exposure to Zone 2 (the City Park area) surface soil was associated with estimated cumulative cancer risks that exceeded the acceptable risk range for residents ( $2.8E-4$ ) but not for recreational persons ( $5.5E-5$ ). For the lifetime resident, the COCs that contributed to cancer risks included arsenic (which was similar to background), Aroclors-1260 and 1254, benz(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, dibenz(a,h)anthracene, and indeno(1,2,3-cd)pyrene.

The target organ-specific HI exceeded 1.0 for the residential child (2.7), but HIs were acceptable for the residential adult, recreational child, and recreational adult. For the residential child, Aroclor-1254 (HQ of 1.7) was the largest contributor to unacceptable noncancer HIs, with additional contributions from cobalt, iron, and antimony. Blood lead concentrations were not predicted to exceed 10 µg/dL for any receptor exposed to Zone 2 surface soil.

#### Eastwick Neighborhood

Exposure to Zone 3 (the Eastwick Neighborhood area) surface soil was associated with estimated cumulative cancer risks that exceeded the acceptable risk range for residents ( $1.9E-4$ ), but not for industrial worker (cancer risk of  $1.2E-5$ ). For the lifetime resident, the COCs that contributed to cancer risks included arsenic (which was similar to background), benz(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, dibenz(a,h)anthracene, indeno(1,2,3-cd)pyrene, Aroclor-1260, and chloroform.

The target organ-specific HI exceeded 1.0 for the residential child (1.2), but HIs were acceptable for the residential adult and industrial worker. For the residential child, the significant contributors to unacceptable noncancer risks included cobalt and iron, which were similar to background, and antimony. Lead was not selected as a COC for Zone 3 surface soil.

#### *Total Soil Risks*

#### Clearview Landfill

Exposure to Zone 1 total soil (representing combined surface and subsurface soil) was associated with estimated cumulative cancer risks that exceeded the acceptable risk range for an industrial worker ( $3.9E-4$ ). The COCs that contributed to cancer risks included arsenic (which was similar to background), Aroclors-1260, 1254, and 1268, dieldrin, heptachlor epoxide, 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) toxicity equivalents (TEQ), benz(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, dibenz(a,h)anthracene, and indeno(1,2,3-cd)pyrene.

For the industrial and construction workers, the maximum of the estimated target organ HIs exceeded 1.0. Antimony was the largest contributor to unacceptable noncancer HIs, with a smaller contribution from heptachlor epoxide. For the construction worker, lead concentrations in fetal blood were predicted to

exceed 10 µg/dL in more than 19 percent of an exposed population, but lead risks were acceptable for the industrial worker. Soil lead concentrations exceeded background.

#### City Park

Exposure to Zone 2 total soil was associated with estimated cumulative cancer risks that exceeded the acceptable risk range for the lifetime resident ( $2.3E-4$ ), but not for the recreational receptors, industrial worker, or construction worker. For the lifetime resident, the COCs that contributed to cancer risks include arsenic (which was similar to background), Aroclors-1260 and 1254, benz(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, dibenz(a,h)anthracene, and indeno(1,2,3-cd)pyrene.

The target organ-specific HI exceeded 1.0 for the residential child (1.2), but HIs were acceptable for the residential adult, recreational child and adult, industrial worker, and construction worker. For the residential child, Aroclor-1254 was the largest contributor to unacceptable noncancer HIs, with additional contributions from cobalt and iron, which were similar to background, and antimony. For the pregnant workers, lead concentrations in fetal blood were predicted to exceed 10 µg/dL in 9.3 percent of an exposed population. Blood lead concentrations were not predicted to exceed 10 µg/dL for any receptors exposed to Zone 2 total soil.

#### Eastwick Neighborhood

Exposure to Zone 3 total soil was associated with estimated cumulative cancer risks that exceeded the acceptable risk range for the lifetime resident ( $1.3E-4$ ), but not for the industrial or construction workers. For the lifetime resident, the COCs that contributed to cancer risks included arsenic (which was similar to background), benz(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, dibenz(a,h)anthracene, indeno(1,2,3-cd)pyrene, and chloroform (for vapor intrusion from soil gas to indoor air).

Noncancer HIs did not exceed 1.0 for residential receptors, the industrial worker, or the construction worker. Lead was not selected as a COC for Zone 3 total soil.

#### *Surface Water and Sediment Risks*

##### Darby and Cobbs Creeks

For the recreational receptors and the construction worker exposed to surface water and sediment from Darby Creek, the estimated cancer risks did not exceed the acceptable risk range. Noncancer HIs for exposure to these creeks did not exceed 1.0 for any receptor. Blood lead modeling was not applicable to surface water and sediment risks because the subject models have not been developed for this type of exposure scenario.

##### Tinicum Marsh

For the recreational receptors and the construction worker exposed to surface water and sediment from Tinicum Marsh, the estimated cancer risks were in the acceptable range. Noncancer HIs for exposure to Tinicum Marsh did not exceed 1.0 for any receptor. Blood lead modeling was not applicable to surface water and sediment risks.

### Leachate Seeps

For the recreational receptors and the construction worker exposed to water and sediment from leachate seeps, the estimated cancer risks did not exceed the acceptable risk range. Noncancer HIs for exposure to leachate seeps did not exceed 1.0 for any receptor. Blood lead modeling was not applicable to surface water and sediment risks.

### *Fish Tissue Risks*

### Darby and Cobbs Creeks

Cancer risks were estimated for the lifetime consumer that recreationally catches fish from Darby and Cobbs Creeks, and for the lifetime receptor that engages in subsistence fishing. The estimated cumulative cancer risks slightly exceeded  $1\text{E-}4$  for lifetime subsistence (cancer risk of  $3.6\text{E-}2$ ) and recreational fishers (cancer risk of  $5.3\text{E-}3$ ). The COCs that contributed significantly to cancer risks included arsenic, PCBs, dichlorodiphenyldichloroethylene (DDE), dichlorodiphenyltrichloroethane (DDT), alpha- and gamma-chlordane, beta-benzene hexachloride (BHC), dieldrin, heptachlor, heptachlor epoxide, benz(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, dibenz(a,h)anthracene, and indeno(1,2,3-cd)pyrene.

For the subsistence fisher and recreational consumers of fish from these creeks, child and adult target organ HIs exceeded 1.0. The COCs contributing to noncancer hazards for these receptors included arsenic, cadmium, chromium, copper lead, mercury, nickel, selenium, silver, alpha- and gamma-chlordane, dieldrin, heptachlor epoxide, DDT, fluoranthene, phenanthrene, and pyrene. Blood lead concentrations were predicted to exceed  $10\text{ }\mu\text{g/dL}$  in 100 percent of individuals exposed, including child and adult subsistence fishers, and recreational consumers of fish.

### Tinicum Marsh

Cancer risks were estimated for the lifetime consumer that recreationally catches fish from Tinicum Marsh, and for the lifetime receptor that engages in subsistence fishing. The estimated cumulative cancer risks for these receptors exceeded the acceptable risk range ( $1\text{E-}4$ ). The COCs that contributed significantly to cancer risks included arsenic, DDE, DDT, alpha- and gamma-chlordane, alpha-BHC, dieldrin, heptachlor epoxide, benz(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, dibenz(a,h)anthracene, and indeno(1,2,3-cd)pyrene.

For the child and adult subsistence fisher and recreational consumers of fish from Tinicum Marsh, target organ HIs exceeded 1.0. The COCs contributing significantly to noncancer hazards for these receptors include arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, zinc, alpha- and gamma-chlordane, dieldrin, heptachlor epoxide, and DDT. Blood lead concentrations were predicted to exceed  $10\text{ }\mu\text{g/dL}$  in 100 percent of individuals exposed, including child and adult subsistence fishers, and recreational consumers of fish.

### *Groundwater Risks*

Exposure to groundwater was associated with estimated cumulative cancer risks that exceeded the acceptable risk range for residents ( $6.6\text{E-}2$ ) and industrial workers ( $2.5\text{E-}4$ ), but not for construction workers.

The major cancer risk drivers for the lifetime resident were 2,3,7,8-TCDD TEQ, arsenic, Arochlor-1260, 1242, and 1248, atrazine, aldrin, dieldrin, heptachlor epoxide, 2,4-dinitrotolulene, 2,6-dinitrotoluene, 3,3'-

dichlorobenzidine, benz(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, dibenz(a,h)anthracene, indeno(1,2,3-cd)pyrene, bis(2-ethylhexyl)phthalate, N-nitroso-di-n-propylamine, PCE, and vinyl chloride. Additional cancer risk drivers displaying risks between 1E-6 and 1E-5 were 4,4'-dichlorodiphenyldichloroethane (DDD), 4,4'-DDE, 4,4'-DDT, alpha- and beta-BHC, delta- and gamma-BHC (lindane), gamma-chlordane, heptachlor, Aroclor-1016, 2,2'-oxybis(1-chloropropane), 3-nitroaniline, 4-nitroaniline, benzene, carbon tetrachloride, chloroform, trichloroethene, naphthalene, and 1,4-dichlorobenzene. Collectively, groundwater exposure to the lifetime resident poses the highest risk among the exposure scenarios assessed for the site.

For the industrial worker exposed to groundwater (i.e., in an excavation trench), the major cancer risk drivers were 2,3,7,8-TCDD TEQs, Aroclor-1260, benzo(a)pyrene, dibenz(a,h)anthracene, benzo(b)fluoranthene, benzo(k)fluoranthene, indeno(1,2,3-cd)pyrene, benzo(a)anthracene, naphthalene, chloroform, 2,2'-oxybis(1-chloropropane), benzene, and 1,4-dichlorobenzene. Cancer risks for the construction worker were based on a much shorter exposure duration compared to the industrial worker; therefore, did not exceed 1E-4.

For the child resident (HI=43) and adult resident (HI=31), the maximum of the estimated target organ HIs exceeded 1.0, but not for construction and industrial workers. For the residential child, the COCs exhibiting HQs greater than 1.0 include 2,3,7,8-TCDD TEQs, Aroclor-1016, aluminum, antimony, arsenic, cadmium, cobalt, copper, iron, manganese, mercury, vanadium, and thallium. Additional COCs contributed to target organ-specific HIs that exceeded 1.0, but individually were associated with HQ contributions of less than 1.0, including zinc, 2,6-dinitrotoluene, 2,4-dinitrotoluene, nitrobenzene, and barium.

Blood lead concentrations were predicted to exceed 10 µg/dL in 100 percent of an exposed population of child residents. Blood lead predictions were not able to be generated for construction workers, but a qualitative evaluation suggests that blood lead levels would be acceptable for a construction worker potentially exposed to incidental ingestion of groundwater (i.e., in an excavation trench).

## **ECOLOGICAL RISK ASSESSMENT (ERA)**

The ecological risk assessment (ERA) was conducted to determine potential impacts to ecological receptors from contaminants detected in site media. Initially, a Screening Level Ecological Risk Assessment (SLERA) was conducted to evaluate potential hazard from chemical concentrations detected in media. The SLERA concluded that risk may exist to lower- and upper-level organisms in the terrestrial, tidal marsh/open water, and non-tidal marsh/open water habitat areas. However, it appeared that there was no unsafe risk through food chain exposure to fish-eating animals from contaminants in tidal riverine surface water; therefore, no bioaccumulation in fish tissue.

Since potentially unacceptable risks to ecological receptors were identified during SLERA, the Baseline Ecological Risk Assessment (BERA) was conducted to further evaluate contaminants of potential concern (COPCs) identified during SLERA. As part of the BERA, toxicity tests on surface water, sediment, and soil samples as well as bioaccumulation tests on sediment and surface soil samples were performed. The following summarizes the BERA findings for each of the assessment endpoints.

### Assessment Endpoint #1: Survival and Growth of Aquatic Plants

Four COPCs (mercury, selenium, thallium, and 4-methylphenol) in creek sediment were determined to have HQs exceeding 1.0, based on maximum sediment concentrations. The HQs of the same four COPCs also exceeded 1.0 for the reference area (Darby and Cobbs Creeks upstream of Clearview Landfill). There were seven COPCs with HQs exceeding 1.0 in Tinicum Marsh and four COPCs with HQs exceeding 1.0

in the impoundment. Four COPCs (endosulfan sulfate, bis[2-ethylhexyl]phthalate, dibenzofuran, and acetone) were not characterized because benchmarks for these COPCs were not available. However, their maximum concentrations and the frequency of detections were similar to those in the reference area.

#### Assessment Endpoint #2: Survival, Growth, and Reproduction of Terrestrial Plants

Risk to the terrestrial plants was characterized at the Landfill-Business Area (a.k.a., southern industrial area), Landfill-Wooded and Shrub Area (Clearview Landfill), and City Park areas containing many COPCs with HQs exceeding 1.0. A total of 27 COPCs for the Landfill-Business Area, 29 COPCs for the Landfill-Wooded and Shrub Area, and 24 COPCs for City Park had HQs exceeding 1.0 based on the maximum soil concentrations. Because no site-specific biological studies were conducted for plants, this assessment endpoint was only evaluated by comparing chemical concentrations to conservative screening levels.

#### Assessment Endpoint #3: Survival, Growth, and Reproduction of Aquatic Invertebrates

Risk to the aquatic invertebrates was indicated based on the toxicity tests, particularly the reduced survival of *H. azteca* at three of the six sediment samples in Darby Creek. Adverse effects to aquatic invertebrates in the *Hyalella azteca* were evident from a decline in survival of *H. azteca* for three sediment samples collected from Darby Creek near and downstream of the landfill. Adverse effects to aquatic invertebrates in the *Chironomus tentans* study was limited to a slight depression of reproduction (total number of eggs produced per female) for *C. tentans* in two sediment samples from Darby Creek compared with the control group reproduction.

#### Assessment Endpoint #4: Survival, Growth, and Reproduction of Terrestrial Invertebrates

Risk to the terrestrial invertebrates was characterized at three areas (Landfill-Business Area, Landfill-Wooded and Shrub Area, and City Park). A total of 19 COPCs for the Landfill-Business Area, 22 COPCs for the Landfill-Wooded and Shrub Area, and 12 COPCs for City Park have HQs exceeding 1.0, based on the maximum soil concentrations. The highest HQs for all of the COPCs occurred within the Landfill-Business Area with the exception of lead and mercury in City Park.

In the *Eisenia fetida* toxicity test, there was no significant difference in survival and growth between the laboratory control sample and the site samples, indicating no significant risk of toxicity to terrestrial invertebrates at the site. In the *Enchytraeus albidus* toxicity test, one site sample (upper northern part of City Park) and the background sample had lower survival after 21 days compared to the laboratory control sample, and three site samples and the background sample had lower reproduction compared to the laboratory control sample. Overall risk to the terrestrial invertebrates was determined by the number of COPCs that exceeded their benchmark values. The Landfill-Business Area and Landfill-Wooded and Shrub Area would be considered to pose greater risk to the invertebrates than would City Park.

#### Assessment Endpoint #5: Survival, Growth, and Reproduction of Fish

Only two chemicals were retained as COPCs in surface water (cadmium and zinc) for the BERA. Risk to aquatic life indicated a HQ of 88 for cadmium, based on maximum water concentration. In the 96-hour static renewal acute fathead minnow test, survival ranged from 85% to 95% for the fathead minnows exposed to two seep water samples and a surface water sample collected just downstream of the seeps, indicating no acute impacts to those fishes.

For the groundwater samples, all of the COPCs had HQ exceeding 1.0 with the exception of two analytes (i.e., endrin ketone and benzo[g,h,i]perylene). This assessment was based on compositing all of the

groundwater samples collected during all sampling events and utilizing the maximum concentrations for the risk calculations. The movement of these COPCs from the groundwater to Darby Creek is not known; therefore, the potential for risk could not be appropriately characterized.

#### Assessment Endpoint #6: Survival, Growth, and Reproduction of Aquatic Feeding Birds and Mammals

For this assessment endpoint, dietary exposure concentrations were modeled using the Lesser scaup and the raccoon as the receptor species. When food chain modeling with more representative life history parameters and site-specific invertebrate tissue samples was conducted for the Lesser scaup, model-calculated risk was driven by dibenzofuran in Darby Creek. Possible risk also existed for the aquatic mammals exposed to mercury, selenium, and dibenzofuran in the upstream reference area, and lead in the leachate seeps. When this modeling was conducted for the raccoon, model-calculated risk existed for the aquatic mammals exposed to selenium and dibenzofuran in Darby Creek and the upstream reference area, and nickel and selenium in the leachate seeps.

#### Assessment Endpoint #7: Survival, Growth, and Reproduction of Terrestrial Feeding Birds and Mammals

For this assessment endpoint, dietary exposure concentrations were modeled using the American robin and the short-tailed shrew as the receptor species. When food chain modeling with more representative life history parameters and site-specific invertebrate tissue samples was conducted for the American robin, risk to terrestrial birds was driven by exposure to lead and mercury at all three areas of interest. Possible risk was driven by chromium and vanadium at all three areas of interest, copper at the Landfill-Business Area, and Aroclor-1260 at the Landfill-Wooded and Shrub Area.

When this modeling was conducted for the shrew, risk to the terrestrial mammals was driven by exposures to antimony, cadmium, mercury, and nickel at the Landfill-Business Area; mercury, nickel, and Aroclor-1260 at the Wooded and Shrub Area; and lead and mercury at City Park. There were a number of COPCs posing model-calculated risk to the terrestrial mammals in all three areas of interest.

## **1.0 INTRODUCTION**

### **1.1 Purpose of Report**

The United States Environmental Protection Agency (USEPA) tasked Tetra Tech to perform a Remedial Investigation (RI) at the Lower Darby Creek Area (LDCA) site located in Philadelphia and Delaware Counties, Pennsylvania. The purpose of the RI is to meet the requirements of the USEPA Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), as amended by the Superfund Amendments and Reauthorization Act (SARA). The LDCA site was placed on the final National Priorities List (NPL) in June 2001 due to its potential release of hazardous substances to the nearby surface water, posing a threat to human health, ecological receptors, and other sensitive environments.

The LDCA site consists of two separate landfill sites and is divided into two operable units for administrative purposes as follows:

- Operable Unit 1 (OU1): Clearview Landfill
- Operable Unit 2 (OU2): Folcroft Landfill and Annex

USEPA is the lead agency conducting RI activities for Clearview Landfill (OU1), and a group of potential responsible parties (PRPs) are responsible for performing RI activities for the Folcroft Landfill and Annex (OU2). Accordingly, this RI report was prepared to present information pertaining to the Clearview Landfill site only. Investigation of the Folcroft Landfill and Annex site is beyond the scope of this RI; however, OU2 will be discussed when necessary due to its potential contribution to contamination downstream of the Clearview Landfill site.

The primary objectives of the RI were as follows:

- Characterize the site conditions;
- Determine the nature and extent of contamination;
- Evaluate potential migration pathways of contaminants pertaining to the site;
- Assess potential risks posed by the site to human health, the environment, and ecological receptors
- Develop information to evaluate potential environmental response clean-up options for OU1.

This RI report consists of six volumes - Volume I (the report), and Volumes II through VII (the appendices), and Volume I is organized in the following order:

- Section 1 presents an introduction and site background information.
- Section 2 contains information pertaining to the physical setting.
- Section 3 describes the field investigation activities.
- Section 4 describes the analytical results and discussion of the field investigations.
- Section 5 describes the evaluation of contaminant fate and transport.
- Section 6 presents findings of the human health and ecological risk assessments.
- Section 7 lists references.

### **1.2 Site Background**

The LDCA site is located north of the Philadelphia International Airport (Figure 1-1), and in an industrialized portion of southeastern Delaware and southwestern Philadelphia Counties in Pennsylvania



(Figure 1-2). There are several creeks near the LDCA site, including Hermesprota, Cobbs, Darby, and Thoroughfare. These creeks generally flow from north to south and discharge into the Delaware River approximately 2.5 miles downstream of the LDCA site.

When the LDCA site was originally proposed for placement on the NPL on May 11, 2000, the following six contiguous properties located on both sides of Darby Creek were included as potential sources of contamination at the site: (1) Clearview Landfill; (2) Industrial Drive properties; (3) Sun Oil Darby Creek Tank Farm (includes the Catalyst Disposal and the Oily Sludge Disposal Areas); (4) former Delaware County Sewage Treatment Plant; (5) former Delaware County Incinerator; and (6) Folcroft Landfill and Annex (Figure 1-2).

However, after reviewing the public comments received, the USEPA promulgated the LDCA site as two separate landfills: the Clearview Landfill, and the Folcroft Landfill and Annex. Therefore, only these two landfills were formally included as sources of contamination when the LDCA site was placed on the final NPL on June 14, 2001 (USEPA, 2001).

In 2006, the USEPA finalized a legal agreement with a group of 14 PRPs to perform the RI/Feasibility Study (FS) for the Folcroft Landfill and Annex site. After the site was listed in NPL, the USEPA became the lead agency responsible for conducting CERCLA-related investigations for the Clearview Landfill. A group of PRPs are leading the investigation of the Folcroft Landfill and Annex with USEPA's oversight and in coordination with the U. S. Fish and Wildlife Service (USFWS), which owns the Folcroft Landfill.

### 1.3 Site History

The two landfill sites, the Clearview Landfill and the Folcroft Landfill and Annex, were determined to be primary sources of contamination at the LDCA site. However, there were other probable sources that might affect portions of Darby Creek that includes fisheries, wetlands, and other sensitive environments such as the John Heinz National Wildlife Refuge (NWR) at Tinicum, which is the largest remaining freshwater tidal marsh in Pennsylvania. A general description of each source, including types of hazardous materials present, historical activities, and prior investigations and response actions, is provided below.

#### 1.3.1 Clearview Landfill (OU1)

The Clearview Landfill is located along the eastern bank of Darby and Cobbs Creeks, at 83<sup>rd</sup> Street and Buist Avenue. The landfill footprint currently resides partly in Delaware County and Philadelphia County (Figure 1-2), and includes the Clearview Landfill, the City Park east of the landfill, and the Eastwick neighborhood. The administrative boundary of the Clearview Landfill is not clearly defined because former landfilling operations, which initially began on a Delaware County land parcel, spilled over onto property located within Philadelphia County (the City of Philadelphia) limits. During the mid-1970s when development began on the Eastwick residential neighborhood (see Figure 1-3), a considerable amount of waste was excavated and moved from the City of Philadelphia portion of the site to the Delaware County portion, where excavated materials were subsequently placed, graded and partially covered with fill. As a result, the present areal extent of the Clearview Landfill lies almost entirely within Delaware County, and the City Park and Eastwick neighborhood lie within Philadelphia County.

The Clearview Landfill was privately owned and operated without a permit from the 1950s to the 1970s by the Clearview Land Development Corporation, and used for the disposal of municipal and industrial waste collected from the City of Philadelphia and portions of Delaware County (USEPA, 2000). There was no documentation of an engineered cover or functioning run-on/runoff control system installed at Clearview Landfill. In addition, there were no records of the types and volume of waste materials accepted at the landfill during operation.

The USEPA's Environmental Photographic Interpretation Center (EPIC), Environmental Monitoring System Laboratory completed an analysis of the Clearview Landfill (USEPA, 1984). As shown in historical aerial photographs of the Clearview Landfill (Figure 1-3), trash disposal at the landfill commenced in the early 1950s. The 1953 aerial photograph showed a 3.3-acre area with debris and earthen mounds north and south of an access road leading into the landfill from Buist Avenue. It also showed that the landfill was situated on and surrounded by wetlands, and several small unnamed streams were present north and west of the landfill. In addition, junked vehicles, debris, and dark-toned material were visible east of the landfill along Buist Avenue.

The 1965 aerial photograph indicated that Clearview Landfill had significantly expanded and covered approximately 55.5 acres with substantial filling activities in the northern and eastern portions of the landfill. The former wetlands and streams have been filled, altering their courses to flow along the eastern border of the landfill, south to Darby Creek. Junked autos and debris were visible in the northwest corner of the landfill. A large pile of dark-toned material, a deep pit containing dark standing liquid, and a crane were clearly evident in this area. Numerous vehicles associated with landfilling activities (e.g., trash trucks, dump trucks, and earthmoving equipment) were also present on-site.

As depicted in an aerial photograph taken in 1973 when Clearview Landfill was closed, the landfill had expanded to the east and bordered Buist Avenue, and covered approximately 65 acres. The stream formerly located east of the landfill was filled. Pools of standing liquid and pits containing liquid (the constituents of the liquid were not determined) were observed on the landfill surface. Tank cars (tanks) and dark stains were also observed on the landfill, indicating that liquid waste may have been brought to the landfill. A new access road led directly from 84<sup>th</sup> Street to the northwest corner of the landfill. Construction of new structure (i.e., garage) had begun at the southern end of the landfill. The 1973 aerial photograph also revealed that new residential properties had been constructed east of the landfill.

The 1983 aerial photograph showed new residential properties constructed at the southeast corner of the landfill, possibly on top of a formerly filled area that had been visible in the 1953 aerial photograph. Construction of the recreation area at the northeast end of the landfill had been completed. A well-traveled access road extended north from the active area in the southern portion of the landfill. Debris was scattered on both sides of the access road.

Historical aerial photographs clearly indicate that former wetland areas located along and adjacent to the Delaware County and City of Philadelphia boundary line were filled and overlain by a thick layer of waste materials during landfill operations. These former wetland areas now appear topographically flat since waste materials were moved into the current landfill area.

The 2002 aerial photograph showed a recent view of the landfill. Currently, several businesses are being operated at the southern end of Clearview Landfill in an area referred to as the Southern Industrial Area. City Wide Waste Disposal Services operates a trash hauling business and stores trucks on-site. Other on-site businesses appear to involve a white-goods (appliances) recycling operation, drum storing and recycling, auto repair and salvage, and truck/equipment storing and snow plowing operation. Additional

ad-hoc businesses also exist on-site. Evidence of frequent illegal trash and waste dumping is also present at the site.

#### Post-Closure Operations at Clearview Landfill and State Actions

In August 1973, due to several violations of state regulations (related to land disposal) and the absence of a landfill permit, the Pennsylvania Department of Environmental Resources (PADER), which is now the Pennsylvania Department of Environmental Protection (PADEP), took court action against the Clearview Land Development Corporation, and ordered it to cease all waste disposal activities at the landfill and follow a prescribed closure plan. Since the 1973 order, the property continued to be used for other waste disposal operations, as described below.

PADEP granted ROMA Associates, Inc. a permit to construct and operate a batch asphalt plant at the southern portion of the landfill in 1973. This plant was operated from 1973 to 1976.

In 1976, the Philadelphia Redevelopment Authority (PRA) covered and seeded a portion of the landfill. However, further information on the type of a cover and its location was not available.

In June 1980, PADEP conducted an investigation of the reported dumping and open burning of waste materials at the Clearview Landfill. During the investigation, it was found that waste materials, including demolition waste, tires, furniture, household appliances, and mattresses, had been disposed of at the top and on the southwestern face of the landfill. In addition, an area on the east bank of Cobbs Creek was being filled with significant amount of lumber, rugs, and other waste materials. PADEP issued a Notice of Violation (NOV) to the owner (Mr. Richard Heller) of Clearview Land Development Corporation.

In 1980 and 1981, Graves Resource Management (GRM) operated an unpermitted hazardous waste transfer, storage, and disposal facility in the southern part of the landfill near the bank of Darby Creek. A PADEP Order dated November 19, 1981, found that GRM continued to operate the facility in violation of Section 403 of the Pennsylvania Solid Waste Management Act (PASWMA). In this Order, numerous violations were cited, including acceptance of waste from unlicensed haulers, failure to maintain the facility so as to minimize the possibility of release of waste to the environment, the presence of unclosed containers during storage at the facility, and no management to prevent leaks and spills of waste. Due to these violations, the facility's interim status was revoked (USEPA, 2000). In 1984, the owner of the operations, Mr. Albert F. Ingram, was sentenced to a prison term for two counts of transporting and dumping hazardous waste committed in 1982.

In November 1981, PADEP conducted another inspection of Clearview Landfill, and noted several dump areas on the landfill with a large quantity of demolition debris, old car parts, bulky items, trash, tires, granular insulation, and black ash. In addition, large storage containers owned by GRM were present. Since these conditions were in direct violation of the August 1973 court-ordered closure of the landfill, PADEP ordered closure of the GRM facility and issued another NOV to Mr. Heller, with regard to violations of the PASWMA and several sections of PADEP's rules and regulations.

A PADEP site inspection (SI) in September 1982 reported that a leachate pond was present on the landfill near Darby Creek, and a foul odor was detected in the stream.

In October 1982, DeLorenzo applied for a permit to operate a solid waste transfer station at Clearview Landfill. In the late 1980s, three other companies (DeLorenzo Twin County Disposal, Bizarro Corporation, and Eagleville Excavating) were located at the landfill and operated by Mr. Heller (the owner of Clearview Land Development Corporation).

In December 1982, PADEP conducted an inspection of Clearview Landfill and noted that waste materials, including demolition debris, abandoned automobiles and parts, and scrap metals, had been deposited directly on the ground. PADEP issued a third NOV to Mr. Heller regarding violations of the PASWMA, and several sections of PADEP's rules and regulations.

In 1984, a former employee of Clearview Land Development Corporation testified in court that he had helped bury chemicals at the landfill while he was employed from 1966 to 1973.

In May 1984, December 1986, and October 1987, PADEP sent Mr. Heller additional NOVs for numerous violations of the PASWMA.

According to the PADEP complaint of equity against Mr. Heller, illegal disposal of waste at the landfill continued at least until 1998, as evidenced by waste deposition observed at the landfill.

#### EPA Actions

EPA Region III's Field Investigation Team (FIT3) conducted site investigations in 1983 and 1984, and collected surface water, sediment, soil, and leachate samples from the landfill and Darby Creek. The analytical results indicated that PCBs and PAHs were detected in leachate samples, PAHs in soil samples, and PCBs in both stream and soil samples.

In September 1990, the FIT3 observed areas of recent dumping throughout the landfill and three leachate seeps draining into Darby Creek on the western edge of the landfill. Contaminants such as VOCs, semivolatile organic compounds (SVOCs), and PAHs were detected in the leachate seeps and downstream sediments.

In May 1998, USEPA conducted an area-wide investigation at the LDCA site to identify possible threats to human health and the environment posed by waste sources along Darby Creek, and to determine the placement of the LDCA site on the NPL. During this investigation, signs of erosion on the landfill cover along the creek banks, exposed debris piles, and leachate seeps were observed. Soil and waste samples collected showed contamination with elevated levels of heavy metals, PAHs, and PCBs.

#### 1.3.2 Folcroft Landfill and Annex (OU2)

The Folcroft Landfill and Annex is located within the John Heinz NWR at Tinicum (see Figure 1-2). The refuge was established in 1972 as the Tinicum National Environmental Center to preserve the largest remaining freshwater tidal marsh in Pennsylvania, and to protect diverse fish and wildlife habitats. In 1980, Congress authorized the U. S. Department of Interior (DOI) to purchase the 62-acre Folcroft Landfill and Annex, to expand the size of the refuge.

The Folcroft Landfill is bordered by Darby Creek on the east, Thoroughfare Creek (a branch of Darby Creek) on the southeast, Hermesprota Creek on the west, the closed Delaware County Incinerator and Delaware County Sewage Treatment plant on the north, and a tidal marsh on the southwest. The Annex is bordered by Hermesprota Creek on the east, a business park to the west, residential developments to the north, and the tidal marsh on the south (Figure 1-2).

The Folcroft Landfill and Annex was owned and operated by Mr. Wilbur C. Henderson (Henderson-Columbia Corporation) from 1959 to 1974. This landfill was originally permitted solely for municipal waste (PADER Solid Waste Permit Number 10053); however, it reportedly received municipal,

industrial, and hospital wastes, as well as incinerator ash and sewage sludge. In addition, the historical aerial photographic analysis indicates that disposal activities took place as early as 1953 (USEPA, 1984). By 1958, the landfill covered approximately 2 acres. The Folcroft Landfill was expanded to approximately 47.5 acres between 1958 and 1971. Landfilling operations extended to Annex in 1971 and covered an additional 16.5 acres. Wastes were placed in wetland areas of the landfill along the edges of Darby, Hermesprota, and Thoroughfare Creeks.

Due to numerous permit violations and improper management, the landfill was closed in 1973. Closing activities included regrading the landfill, reducing the steep slopes, placing a cover, and seeding. Cover materials were dredged soil received from the construction of Interstate 95 and the Sun Oil Company refinery in Marcus Hook, Pennsylvania. A site inspection (USEPA, 2001) reported lack of vegetative cover on the eastern half of the Folcroft Landfill.

Numerous investigations have been conducted at the Folcroft Landfill and Annex. Some of the findings from these inspections are briefly described herein:

- Inspections of the Folcroft Landfill were performed between 1969 and 1973. During these inspections, it was found that the landfill received wastes from the Philadelphia Navy Yard, the Boeing Vertical Company, and the American Viscose Company. It was also found that oily sludge was being disposed of in the southern portion of the landfill; sewage sludge had been dumped on the east side of the landfill; refuse was being pushed directly into a swamp and adjacent water body on the east side of the landfill; and industrial wastes with oily material were being disposed of on the surface of the landfill. In addition, PADEP found a number of leaking drums with liquid flowing toward Tinicum Marsh, and analyzed aqueous and waste samples collected from the Folcroft Landfill. Results from the sample analyses indicated elevated levels of heavy metals such as cadmium, copper, chromium, nickel, zinc, and lead.
- A 1980 site inspection conducted by USEPA identified the following wastes or waste constituents that have been disposed of in the landfill: oily waste; halogenated solvents; aromatic compounds; pesticides; metals; fly ash; asbestos; radioactive materials; municipal waste; hospital waste; and demolition waste.
- In July 1983, USEPA Region III was notified of a fire at the Folcroft Landfill Annex, allegedly caused by the catalytic converter of a vehicle parked over underbrush on the landfill. USEPA implemented an immediate removal action and removed a number of drums located at the landfill. During the removal action, drum and soil samples were also collected and analyzed. The contents of the drums were described in a hazardous manifest, as follows: (1) resin – flammable solid, 170 gallons, waste code D001 (a solid waste that exhibits the characteristic of ignitibility); (2) flammable solids – flammable solid, 85 gallons, waste code D001; (3) water soluble lead – water soluble lead, 170 gallons, waste code D008 (lead concentration higher than 5 mg/L); and (4) asphalt – combustible solid, 85 gallons, waste code D001. In addition, a large quantity of illegally dumped hospital wastes was discovered throughout the surface of the landfill. USEPA covered portion of the landfill with 6 to 8 inches of fly ash (filter cake) supplied by the Philadelphia Electric Company, followed by 12 inches of compacted soil and hydroseeding.
- Upon DOI's acquisition of the 62-acre Folcroft Landfill and Annex in 1980, USEPA, in coordination with USFWS and DOI, investigated contamination in the landfill in 1986. The investigation concluded that the Folcroft Landfill was a source of heavy metals such as aluminum, cyanide, chromium, copper, and nickel in the John Heinz NWR (USEPA/USDOI/USFWS, 1986).

- In a follow-up site investigation, USEPA and USFWS conducted a joint SI in 1988, including the collection of soil, sediment, surface water, seep, and installation/sampling of five groundwater monitoring wells. Three wells (MW-1, 2, and 3) were installed at the toe of Folcroft Landfill along a bermed area outside the fill area, a downgradient well (MW-5) was installed at the Folcroft Annex, and an upgradient well (MW-4) was installed near the former Delaware County Incinerator (Gannet Fleming, 1984). The analytical results indicated that groundwater samples collected from the monitoring wells contained metals and VOCs at elevated concentrations, and the surface soil samples contained heavy metals and PAHs at concentrations equal to or higher than three times the background level.
- Additional investigation was conducted by USEPA in May 1998. During this investigation, several springs and seeps were observed on the southeastern edge of the Folcroft Landfill along Thoroughfare Creek (a tributary of Darby Creek – see Figure 1-2). Signs of erosion on the landfill cover, exposed waste materials, and leachate seeps were also observed. The extent of erosion was most significant along the steeply sloped southern side, nearest to Thoroughfare Creek and the tidal marsh. Groundwater samples were found to be contaminated with heavy metals and VOCs, while soil samples were contaminated with heavy metals at levels equal to or greater than three times the background concentrations.

### 1.3.3 Other Sources of Contamination

When the LDCA site was originally proposed for placement on the NPL, four other sources described previously (Figure 1-2) were also evaluated in the Hazard Ranking System (HRS) Documentation Record, but these sources were not included in the final NPL. In spite of their exclusion as sources of contamination at the LDCA site in the final NPL, the USEPA's sampling results indicate that contaminants detected in Darby Creek are likely associated with all of the aforementioned sources. These four sources are discussed briefly herein, in order from upstream to downstream along Darby Creek.

1. **The Industrial Drive** is a short street that runs southwest from S. 84<sup>th</sup> Street (Hook Rd.) and parallel to Darby Creek. The properties along this street were used as an open dump in the early 1950s. Historical aerial photos (USEPA, 1984) indicated that the area was later utilized for various commercial and industrial purposes, and is currently occupied by salvage yards and a vehicle repair shop. Several sampling events in 1998 and 2000 documented the presence of elevated levels of heavy metals and PAHs in soil (Tetra Tech, 2000).
2. **The Sun Oil Darby Creek Tank Farm** is a crude oil tank storage facility, which is still in operation. It includes three potential contaminant sources, such as the Oily Sludge Disposal Area, the Catalyst Disposal Area, and the Neutralized Hydrofluoric Acid Trash Disposal Area, as per the HRS Documentation Record. This tank farm was constructed by the former Gulf Oil Refinery on a former rock quarry in the late 1940s and early 1950s. It was purchased in 1994 and is currently owned by the Sun Oil Company. This site was used primarily for the disposal of waste materials generated from the Gulf Oil refinery, including oily sludge, various refinery catalysts, scrap metals, and neutralized acid waste. Sampling performed by USEPA showed that groundwater in this property was contaminated with heavy metals and benzene. USEPA also observed a thick oily substance overlying groundwater. In addition, a pipeline leakage incident in 1999/2000 released more than 90,000 gallons of crude oil into the refuge impoundment covering an approximately two-acre area.

3. **The former Delaware County Sewage Treatment Plant** discharged treated water directly to Darby Creek until the early 1970s. Sewage sludge taken from the drying beds was disposed of in the sludge disposal area alongside Darby Creek. The sludge was never removed and has become overgrown with vegetation. This plant is currently used as a pumping station by Delaware County and an animal farm. Soil samples collected from this area indicated the presence of elevated levels of heavy metals, PAHs, and PCBs, including Aroclor 1260.
4. **The former Delaware County Incinerator site** was used for the incineration of municipal waste between the mid-1960s and early 1970s. The incinerator was owned and operated by Delaware County, and handled approximately 500 and 800 tons of refuse per day. Some incinerator ash and residue was placed in a 15-acre area located immediately to the south of the incinerator. Subsurface soil sampling of this fill area was conducted by USEPA in 1998, indicating the presence of contaminants such as heavy metals and dioxin. This property is now occupied by the Delaware County Emergency Services Training Center.

Note that the owners of the three properties (the former Delaware County Incinerator, the former Delaware County Sewage Treatment Plant, and the Sun Oil Darby Creek Tank Farm) intend to voluntarily address contamination on their properties under Pennsylvania's Land Recycling Program, Act 2 of 1995 - the Land Recycling and Environmental Remediation Standards Act, and other regulatory programs.

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## **2.0 PHYSICAL CHARACTERISTICS OF STUDY AREA**

This section describes the physical characteristics of the Clearview Landfill and surrounding areas. Information on site geology, soils, surface water hydrology, hydrogeology, human population, land use, and surface features are provided herein.

### **2.1 Demography and Land Use**

The LDCA site is located in an industrial section of Darby Township, Delaware and Philadelphia Counties in Pennsylvania. In 2000, there were approximately 9,600 people in Darby Township with a population density of 6,739 per square mile in a total area of 1.43 square miles (U.S. Census, 2000). Of these, approximately 3,500 people live within a 0.25-mile radius of Clearview Landfill (Figure 2-1). Residential properties are situated east of Clearview Landfill with population density increasing northeast toward Philadelphia. All residential properties in the Eastwick neighborhood are located within the City of Philadelphia boundary, while Clearview Landfill is entirely located in Delaware County.

According to the 2000 census, races in Darby Township were comprised of White Non-Hispanic (62.0%); Black (36.4%); Hispanic (0.9%); and two or more other races (0.7%). In Darby Township, the length of stay since moving-in is significantly above state average, and house age is also above state average.

The Clearview Landfill is located geographically closer to the City of Philadelphia. The U.S. Census Bureau (2000) reports that the total population in the city was 1,517,550, including White (45%); African American (43.2%); American Indian and Alaska Native (0.3%); Asian (4.5%); some other race (4.8%); and two or more races (2.2%). The population density was 11,233.6 per square miles. The city has a total area of 142.6 square miles, of which 135.1 square miles is land, and 7.6 square miles is water. The year 2000 census data show that there are about 2,300 people living immediately adjacent to Clearview Landfill and 755 homes in the 6-block area between the landfill, S. 84<sup>th</sup> and S. 78<sup>th</sup> Streets, and Linbergh Blvd.

In general, land use within the study area is urban residential mixed with commercial, industrial, and natural area uses. Figure 2-2 shows the land use reported during the 1983 study of the Clearview Landfill using the Anderson Classification System (USEPA, 1984). According to this classification system, the land uses of Clearview Landfill are shown as Commercial/Light Industrial (162); Vacant Urban Lands (173); and Dump (177), although the entire land has been used for dumping waste. Residential land use is predominant in close proximity to the east of Clearview Landfill.

### **2.2 Surface Features**

The LDCA site is in the small portion of the Coastal Plain Physiographic Province that occurs in Pennsylvania, and is very flat except for Clearview Landfill, and Folcroft Landfill and Annex. As shown in Figure 2-3, Clearview Landfill itself has the largest relief of any nearby land surface, rising to an elevation of over 80 feet above mean sea level (msl). Elevations in the vicinity range from 0 to 40 feet above msl.

For purposes of the RI, the Clearview Landfill includes three investigative sub-areas: the Clearview Landfill, the City Park located adjacent to the landfill, and the Eastwick residential community (Figure 2-3). Clearview Landfill is currently situated within Delaware County. However, the historical footprint of the landfill operation was known to extend to the City Park area, which is deeded in Philadelphia County (or City of Philadelphia). The adjacent City Park is currently a public recreational facility, which includes tennis courts, basketball courts, playgrounds, and walking paths.

In addition, the Clearview Landfill is currently used by several businesses (a small portion of the southern area classified as Commercial/Light Industrial in Figure 2-2), including a waste transfer station, an truck/equipment storing and snow plowing business, an auto repair and salvage operation, and a drum recycling (Figure 2-3). Local residents access the landfill area for walking, all terrain vehicle riding, deer hunting, and other activities. Abandoned cars have also been found at the landfill.

### 2.3 Climate

The study area is primarily maritime – typified by a humid, temperate climate. The average temperature is 72°F in the summer months and 32°F in the winter months. Generally, the weather changes every few days in the winter and spring, and less frequently during summer and fall due to slower atmospheric circulation inherent in maritime climate dynamics. According to the National Oceanic and Atmospheric Administration (NOAA), mean annual precipitation is approximately 48 inches with an average of 13.8 inches in summer months, 11 inches over the fall months, and 23.2 inches in winter months. Occasional local periods of drought have been known to occur, but humid conditions are the norm.

### 2.4 Soils

Soils in the vicinity of the Clearview Landfill have been heavily disturbed through many years of urban land use and are generally described as "Made Land" by the U. S. Department of Agriculture (USDA) Natural Resources Conservation Service (NRCS). Surficial geology in the area is generally unconsolidated sedimentary deposits that consist of gravelly sand with some interbedded clay and silt. In addition, part of the area has been extensively filled with fine-grained sediment, dredge spoils, and flood deposits.

The soil types mapped by NRCS (Web Soil Survey) are depicted in Figure 2-4, and the select map units shown near the Clearview Landfill site are described briefly below:

- Made Land, Sanitary Landfill (Mf) – Soils under this map unit consist primarily of Udorthents, sanitary landfill, and similar soils, and are typically found in areas of cut and fill. The material is typically similar in the subsoil or substratum of adjacent soils. In fill or disposal areas, the soil material has more variable characteristics because it usually consists of varying amounts of material from the subsoil and substratum of nearby soils. Slope is reported to vary from 0 to 15 percent. Soils in Clearview Landfill, Industrial Drive, and northeast of Sun Oil Tank Farm are shown to be this type.
- Urban Land (Ub) – This soil type represents approximately 85% of land in Philadelphia County. Its typical setting includes 0 to 8% slope and parent materials consisting of pavement, buildings and other artificially covered areas. This soil type resides in City Park of the Clearview Landfill site.
- Urban Land – Howell Complex (Uh): this soil type is typically mixed with urban land, and Howell and similar soil. In addition to typical urban land setting, this unit has Howell, of which parent material is unconsolidated sediments residuum, consisting of silt loam, sandy clay loam, and clay. It has 0 to 15% slope. This soil type is dominant in the Eastwick neighborhood east of Clearview Landfill.
- Wehadkee Silt Loam (We) – This type consists primarily of Wehadkee and similar soils. This soil type has 0 to 3% slope and does not drain well. It is typically found in flood plains and profiled as

silt loam, silty clay loam, and stratified clay. This soil type is located in the west side of Darby Creek near Clearview Landfill.

- Glenelg Channery Silt Loam (GeB2) – This soil type is found further west of Darby Creek near Clearview Landfill and west of Industrial Drive. It drains well and has a slope of 3 to 8 percent. Its typical profile has 0 to 8 inches of Channery silt loam; 8 to 29 inches of Channery silt loam; and 29 to 50 inches of very channery loam.

## 2.5 Regional Geology

In general, the surficial geology in the vicinity of the LDCA site has little or no original outcropping surficial geology remaining. A regional geology with chronostratigraphic units is shown in Figure 2-5. The Clearview Landfill is situated on unconsolidated Coastal Plain sediments (Quaternary Trenton Gravel at the surface) overlying bedrock of the Wissahickon Formation. Bedrock was encountered at depths ranging from approximately 18 feet in the Eastwick neighborhood to as deep as about 44 feet at the southern part of the landfill during the RI.

The area is predominantly underlain by the Quaternary Age Trenton Formation in thicknesses of up to 40 feet, which consists primarily of medium- to coarse-grained gravelly sand interstratified with clayey silt and sand layers. Beneath the Trenton Formation lie the Pennsauken and Bridgeton Formations, which consist of cross-bedded, cemented sands with interbedded coarse-grained gravel. These formations have a maximum thickness of 30 feet and are present as outcrops in the general surrounding area, but their existence below the Clearview Landfill is unknown.

Beneath the units described above lie the Cretaceous Age Potomac Group (silts and clays with interbedded sands and some gravel) and the Raritan Formation (containing various clays, sand, and gravel members); however, neither can be confirmed below Clearview Landfill.

The Precambrian Age Wissahickon Schist Formation is present beneath the layers described above. This formation consists primarily of oligoclase-mica schist, a group of metamorphic rocks containing parallel layers of flaky minerals such as mica. Because of the intense folding of this unit, its exact thickness is unknown, but is estimated to range from 8,000 to 10,000 feet. The Wissahickon Formation is present within the study area as outcrops in nearby stream channels, including Darby, Cobbs, and Hermesprota Creek Valleys, and underlies the northern end of the Clearview Landfill. This formation was also identified in both Darby and Cobbs Creek upstream of Clearview Landfill during the RI.

## 2.6 Regional Hydrogeology

In general, groundwater regionally flows from the northeast to the southwest direction toward the Delaware River. As shown in Figure 2-5, the southern portion of the Clearview Landfill is underlain by Trenton Aquifer and the northern part of the landfill is underlain by the Wissahickon Aquifer. Both are water table aquifers (i.e., an aquifer which is not confined under pressure; therefore, the water level in a well is the same as the water table outside the well.) Groundwater gradients are typically low in this type of aquifer (e.g., hydraulic gradient of about 10 feet per mile or less). Prior to the construction of the landfill, groundwater flowed from higher elevations to streams that discharge to the Delaware and Schuylkill Rivers. Wells located in close proximity to rivers and creeks caused localized limited extent flow reversals of the typical hydraulic gradient, with recharge to the groundwater coming from the river, instead of groundwater flowing into the river (Greenman et al., 1961). Fluctuation of the groundwater level also occurs in water wells located near tidal waterways due to effects from tidal water level changes.

According to the USEPA Aquifer Classification System in the *Guidelines for Ground-Water Classification Under the [1984] EPA Ground-Water Protection Strategy, Final Draft* (USEPA, 1986), the aquifers underlying the Clearview Landfill are classified as Class I aquifers due to the presence of the John Heinz NWR within 2 miles of the landfill. As per Section 3.3.1 of the guidelines, the aquifer is classified as Class I when the groundwater is “ecologically vital,” and supports a sensitive ecological system and a unique habitat such as the John Heinz NWR.

## 2.7 Public Water Supply

All known residents in the Delaware and Philadelphia Counties are supplied with potable water by a public water supplier, the AQUA Philadelphia Water Company.

Figure 2-6 shows water wells identified in the vicinity of the LDCA site and their uses of water. No drinking water wells are known to exist on the Pennsylvania side of the Delaware River. On the New Jersey side of the Delaware River, drinking water wells are known to exist for Gibbstown and the Borough of Paulsboro (located 5.5 and 4 miles south of Clearview Landfill, respectively).

## 2.8 Surface Water Hydrology

Surface water features associated with the LDCA site consist of streams and marsh areas, as shown in Figure 2-7. Streams in the area include Darby, Cobbs, and Hermesprota Creeks. The main stem of Darby Creek originates in Easttown Township, Chester County and is joined by a number of tributaries as it flows downstream. Cobbs Creek, the major tributary of Darby Creek, converges with Darby Creek north of Clearview Landfill. Darby Creek is then joined by Hermesprota Creek near marsh area in John Heinz NWR at Tinicum. Water from Darby Creek and the marsh ultimately flows into the Delaware River. The confluence of Darby Creek and the Delaware River is approximately 3.5 miles downstream of Clearview Landfill. An impoundment and tidal wetlands exist within the John Heinz NWR.

Darby, Cobbs, and Hermesprota Creeks were listed as impaired waters under Section 303(d) of the 1972 Clean Water Act. Due to pollution issues, portions of Darby and Cobbs Creeks in the vicinity of the Clearview Landfill were listed as requiring a Total Maximum Daily Load (TMDL), which specifies the maximum amount of a pollutant that a water body can receive, while meeting water quality standards. A small portion of Cobbs Creek is listed as not requiring a TMDL, but still addressed with pollution issues. Portions of Darby and Hermesprota Creeks are listed as being unassessed due to insufficient data.

Tidal influence exists throughout the lower portion of Darby Creek and upstream as far as Clearview Landfill. On average, Darby Creek is tidal up to the confluence of Darby Creek and Cobb Creek, located near the northern portion of the landfill, but the extent of tidal influence changes depending on climate conditions.

Flood plains encroach significantly onto the study area. Figure 2-7 shows the 100-year flood plain line superimposed onto wetlands mapping. As an example of a historical event, Hurricane Floyd in 1999 caused significant flooding throughout the Cobbs and Darby Creeks, including the Eastwick neighborhood and surrounding area, inundating many homes. Based on the mapping of the flood plain, the ground surface elevation, and other information, it appears that flooding could be a more common event than indicated by the 100-year flood plain mapping alone.

## 2.9 Wetlands

The John Heinz NWR consists of approximately 1,200 acres of wetlands within two miles of Darby Creek near the LDCA site (USFWS, 2001). Based on the 2005 National Wetlands Inventory (NWI), wetland classification in the study area is shown in Figure 2-7. Darby Creek and its downstream portion are classified as riverine, tidal, emergent wetlands. Two wetlands, including one near Clearview Landfill and the other near the Sun Oil Tank Farm, are classified as palustrine, emergent wetlands. Additional wetlands that are classified as palustrine, open water/unknown bottom and palustrine, unconsolidated bottom wetlands are located on the west side of Darby Creek near the Sun Oil Tank Farm. In the lower portion of the LDCA site just east of Darby Creek, a 40-acre area is classified as lacustrine, limnetic, open water/unknown bottom wetlands. Additionally, 131 and 132 acres of wetlands areas south of the Folcroft Landfill and Annex are considered riverine, tidal, emergent wetland, and palustrine, emergent, persistent wetlands, respectively. A wetland quality assessment was not performed during the RI to determine the quality rankings of the various wetlands encountered.

## 2.10 Ecology

During the RI, biological and ecological information on the LDCA site was collected with particular emphasis on identifying sensitive environments, endangered species and their habitats, and those species consumed by humans or found in human food chains. A discussion of the ecological risks posed by OU1 is presented in section 6.2. In addition, a full ecological risk assessment report is available in Appendix V.

### 2.10.1 Eco-Region and Physiographic Area

Ecoregions denote areas of general similarity in ecosystems, and in the type, quality, and quantity of environmental resources. A roman numerical hierarchical scheme has been adopted for different levels of ecological regions. Level I is the coarsest level, dividing North America into 15 ecological regions; Level II divides the continent into 52 regions; Level III contains 104 regions in each continent; and Level IV is a further subdivision of Level III ecoregions (USEPA, 2003a).

Figure 2-8 shows Levels III and IV Ecoregions in the USEPA Region 3. The LDCA site is in part of the Delaware River Terraces and Uplands zone (Zone 63a) of the Middle Atlantic Coastal Plain Ecoregion (Level IV). In general, the Level IV Ecoregion is a narrow, marshy, nearly level-to-rolling lowland adjacent to the Delaware River estuary and Delaware Bay that extends from southeastern Pennsylvania to southeastern Delaware. It is characterized by low, nearly level terraces; an ocean modified climate; a long growing season; freshwater inter-tidal marshes; saltwater marshes; and small, sluggish, meandering streams. Low lying areas are commonly saturated or flooded during the growing season. Saline marsh deposits dominate, and alluvial and estuarine sand and silt are also widespread. These deposits are underlain by unconsolidated and easily eroded Quaternary gravels, sands, and silts. Elevations are less than 60 feet, and local relief is less than 35 feet. Streams have low gradients and are tidally influenced (USEPA, 2003a). Note that the Delaware River is saline up to approximately river mile 93 (near the Walt Whitman Bridge) from the mouth of the Delaware River near the Atlantic Ocean.

### 2.10.2 Threatened and/or Endangered Species of Concern

During the RI, investigation on the presence of threatened and/or endangered species at the LDCA site was conducted by contacting several state and federal agencies, including USFWS, Pennsylvania Department of Conservation and Natural Resources (PADCNR) Bureau of Forestry, Pennsylvania Fish and Boat Commission (PFBC), and the Pennsylvania Game Commission (PGC). The response letters from these agencies are summarized below:

Federally Listed Species: The USFWS reported that there are no federally listed or proposed threatened and/or endangered species under their jurisdiction, based on a review of the LDCA site location (USFWS, 2010).

State-Listed Plant Species: The PADCNr (2010) reported that potential impact of the project to plant, terrestrial invertebrates, natural communities, and geologic features of concern under DCNR's responsibility is anticipated. There are plant species of special concern within the LDCA site, including waterhemp ragweed (*Amaranthus cannabinus*) in uppermost zone of freshwater intertidal marsh; field dodder (*Cuscuta pentagona*), which is parasitic on many hosts, in old field, thickets and wet banks; Walter's barnyard grass (*Echinochloa walteri*) in tidal marshes and mudflats; *Eupatorium rotundifolium* in sandy or clayey fields and open thickets; forked rush (*Juncus dichotomus*) in moist, sandy old field, open woods and gravel pits; velvety panic-grass (*Panicum scoparium*) in moist meadows and swales; shrubby camphor weed (*Pluchea odorata*) in tidal wetlands, wet ditches, and railroad ballasts; river bulrush (*Schoenoplectus fluviatilis*) in moist sandy shores, and tidal and non-tidal marshes; wild senna (*Senna marilandica*) in dry roadsides and thickets; and Indian wild rice (*Zizania aquatica*) in tidal and non-tidal marshes.

State-Listed Animal Species: The PFBC reported that several state-listed rare or protected animal species are known to exist in the vicinity of the site according to their records and the Pennsylvania Natural Diversity Inventory (PNDI) database (PFBC, 2010). The red-bellied turtle (*Pseudemys rubriventris*) is a threatened species, while the Coastal plain leopard frog (*Rana utricularia*), New Jersey chorus frog (*Pseudacris feriarum kalmi*), and threespine stickleback (*Gasterosteus aculeatus*) are endangered species expected to occur in the vicinity of the site. The eastern mudminnow (*Umbra pygmaea*) is a candidate species.

The red-bellied turtle is one of the largest native aquatic turtles in Pennsylvania (PADCNr, 2010). Relatively large, deep creeks, rivers, ponds, lakes and marshes with ample basking sites are preferred habitats by these turtles. These turtles are restricted to the southcentral and southeastern regions of Pennsylvania. The existence of this species is threatened by habitat destruction, poor water quality, and competition with aggressive non-native turtle species (e.g., red-eared slider, *Trachemys scripta elegans*) that share its range and habitat. Red-bellied turtle presence within the project area is well-documented, and studies gathering more information on their habitat uses and needs are on-going.

The coastal plain leopard frog (a.k.a. southern leopard frog) resembles the northern leopard frog, but has a white spot in the center of its eardrum, fewer dark spot on its sides, and a longer, pointed head than the northern leopard frog. It lives and breeds in shallow, freshwater habitats and slightly brackish coastal marshes in southeastern Pennsylvania. The coastal plain leopard frog is endangered primarily because of habitat loss from development and industrial activities (PFBC, 2010).

The New Jersey chorus frog is a subspecies of the western chorus frog. It breeds from February to June in small relatively open bodies of water with a mixture of shrubby and herbaceous aquatic vegetation, or sometimes in the shallow backwater areas of larger bodies of water with similar vegetation. The small populations of this species are threatened by pollution and filling/clearing of wetlands and breeding habitat (PADCNr, 2010).

The threespine stickleback is a small, schooling fish, which is the rarest of the Pennsylvania fish species. It prefers clear water that is quite and weedy. This species is known to live only in the lower Delaware River Estuary and some of its tributaries (PFBC, 2010). This species is endangered due to habitat destruction and water pollution (PADCNr, 2010).

The eastern mudminnow is highly secretive, and inhabits very shallow water under vegetation and debris within marshes, weedy shores of lakes, or stagnant streams within the Delaware River drainage. This species is rare due to habitat destruction and water pollution.

State-Listed Species of Concern: The PGC (2004) also reported that the LDCA site contains potential habitat for several state-listed avian endangered and/or threatened species. These species include the following: great egret (*Ardea alba*), black-crowned night heron (*Nycticorax nycticorax*), yellow-crowned night heron (*Nycticorax violaceus*), short-eared owl (*Asio flammeus*), king rail (*Rallus elegans*), least bittern (*Ixobrychus exilis*), American bittern (*Botaurus lentiginosus*), and sedge wren (*Cistothorus platensis*).

#### 2.10.3 Sensitive Environments

John Heinz NWR (the Refuge) at Tinicum is the primary sensitive environment in the vicinity of Clearview Landfill. The Refuge existed prior to the first settlement in the region in 1634 (USFWS, 2003), and is home to a variety of wildlife. Birdwatchers have recorded more than 280 species of birds in and around the Refuge. The Refuge is one of the few places in Pennsylvania where the state-endangered red-bellied turtle and southern leopard frog can be found. It is located on the Atlantic Coastal Plain, which is only present in the extreme southeastern corner of the state and has been highly impacted by industrial activity. Other sensitive areas include wetland areas along the banks and flood plains of Darby Creek.

In addition, Clearview Landfill is located on a major waterfowl migration route that is part of the Atlantic Flyway. Numerous waterfowls are observed on the waterways near Clearview Landfill. Within the general study area, wetlands that serve as resting areas for migrating waterfowl are located in the Refuge. Water from upstream areas of the City of Philadelphia and portions of Delaware County eventually enters the Refuge via Darby Creek. Moreover, Cobbs and Darby Creeks are listed as warm-water fishing streams by the Pennsylvania Fish and Boat Commission (PFBC).

#### 2.10.4 Habitat Quality

Although the LDCA site is located within a highly urbanized industrial setting that could negatively affect habitat quality in the study area, there are several significant and unique habitat areas adjacent to the site. The John Heinz NWR at Tinicum is an approximately 200-acre wetland that represents the largest freshwater tidal marsh remaining in Pennsylvania. Additional information on the Refuge, including a list of animal and plant species can be found at <http://www.fws.gov/heinz/index.html>.

Point source and non-point source pollution within the Darby Creek watershed affect water quality and available food-chain support for wildlife, including bird species within the Refuge. Growing industrialization continues to encroach on the Refuge, resulting in reduced open space and increased pressure on Refuge lands. Human activities (e.g., recreational use and water level control) also affect the quality of the habitat for nesting birds in the Refuge.

The 145-acre shallow impoundment in the John Heinz NWR is connected to part of the tidal marsh via a tidal gate. This large open water along with the adjacent heavily vegetated tidal wetland form an ideal habitat for migratory waterfowl. The impoundment also contains a large population of common carp. Carp foraging increases the turbidity of the water and uproots aquatic vegetation. The increased turbidity covers the vegetation at the bottom of the impoundment with silt and eventually kills them. The Canada Geese which are also present in the area and carp have destroyed areas newly planted with wild rice, which historically dominated the marsh.

Invasive species are abundant in the watershed as well as the Refuge. The extent of the phragmites is of great concern. Other invasive species that limit the productivity of the emergent wetlands and that require management and control include spatterdock and purple loosestrife. Japanese knotweed, lesser celandine, Asian tearthumb, Japanese honeysuckle, mile-a-minute, bittersweet, Paulownia, kudzu, and Ailanthus are also pervasive in the upland habitat of the Refuge.

The habitat quality around Clearview Landfill is considered poor because of constant disturbance, poor natural food sources (except for scavengers), and lack of quality cover on Made-Land soils, all resulting from urbanization. Cobbs Creek upstream of Clearview Landfill has severely eroded. Based on field observations, it appeared that the general habitat quality within/near the Clearview Landfill is of poor quality. However, the on-site wooded area along with the riparian corridor does represent habitat that is unique to the urbanized setting, and its habitat quality is better than that in the surrounding residential area.



**2.0      PHYSICAL CHARACTERISTICS OF STUDY AREA 1**

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### 3.0 STUDY AREA INVESTIGATION

The RI field investigation was designed to obtain data and improve the understanding of the site conditions. This section describes the sampling and field work activities conducted during the RI. More detailed information is also available in the RI Work Plan (Tetra Tech, 2002a; 2002b; and 2002c).

The field investigation included the following major elements in chronological order:

- Aerial mapping, surveying, and geographic information system (GIS) within the study area (Spring 2002).
- Sediment and surface water sampling in Cobbs and Darby Creeks, upstream and downstream of Clearview Landfill (Spring/Summer 2002).
- Leachate seep sampling from the bank of Clearview Landfill along Darby Creek (Summer 2002).
- Installation of soil borings in City Park and Eastwick neighborhood, and monitoring wells in City Park, followed by soil, soil/landfill gas, and groundwater sample collection (first round - Winter/Spring 2003).
- Installation of staff gages along Darby and Cobbs Creeks for the tidal study (January 2003).
- Sampling of stormwater and ponded water in City Park (Spring 2003).
- Installation of additional soil borings in City Park and Eastwick neighborhood, and permanent vapor monitoring wells in City Park, followed by soil, soil/landfill gas, and groundwater sample collection (second round - summer 2004).
- Collection of earthworm, and sediment and surface water sampling performed in City Park, and Darby and Cobbs Creeks for ecological assessment (Fall 2005).
- Installation of soil borings and groundwater monitoring wells in Clearview Landfill (Winter 2005/2006). Samples of soil, soil/landfill gas, and groundwater were collected.
- Landfill leachate seep sampling from the banks of Clearview Landfill along Darby Creek (Spring 2006).
- Groundwater sampling from all permanent monitoring wells (second round – summer 2006).

#### 3.1 Mapping and Surveying

A comprehensive aerial base map of the approximate four-square mile study area was developed to identify major site features and determine locations of sampling activities. The base map included the following four basic components:

- Vector-format drawings of the study area to determine topography, elevations, and utilities locations to help further understand physical characteristics of the study area.
- Aerial photographs of the study area, consisting of seamless digital aerial photos to orient all data to site features.
- Site-wide geographical information system (GIS) to allow rapid storage/mapping/integration of site maps to digital products such as analytical results.
- Stream channel profiling by global positioning system (GPS) survey methods for sediment/erosion and surface water/sediment analyses.

##### 3.1.1 Aerial Base Mapping

Aerial base mapping was performed by Land and Mapping Services, Inc. in April 2001, and prepared with a scale of 1 inch equals 200 feet and a contour interval of 2 feet. The visual aerial photography component of the surveying task was implemented in early spring when vegetation was minimal.

Pertinent features, including roads, waterways, stormwater drainages, wooded areas, major structures and obstructions, and utilities, were identified; however, property boundaries were not identified during this task.

The base map was projected using the Universal Transverse Mercator (UTM) coordinate system which was originally developed by the North Atlantic Treaty Organization (NATO) in 1947 (the term "map projection" actually means any mathematical transformation of the globe onto some other surface.) The UTM longitude grid covers the 48 contiguous states in the U.S. with 10 zones – from Zone 10 on the West coast through Zone 19 in New England. The study area is located within Zone 18. For horizontal and vertical location control, the North American Datum of 1983 (NAD83) and National Geodetic Vertical Datum of 1988 (NGVD88) were used for horizontal and vertical datum, respectively. In addition to the aerial photography for mapping, raster-format orthographically-corrected aerial photograph tiles of the entire study area were prepared for subsequent use in ArcGIS 9.1, AutoCAD, modeling, site analysis, and other mapping-related activities. The resulting aerial base mapping in a 500-foot grid system (grids are not shown) is shown in Figure 3-1 and also provided electronically in Appendix A.

### 3.1.2 Surveying

Surveying activities were performed to horizontally and vertically locate various site features. Ground surveying, utility surveying, storm sewers, and stream channel profiling were conducted for this purpose.

Ground surveying was conducted for selected site features where point-based data (e.g., coordinates) were needed or aerial mapping was obscured. Surveying on elevations and locations of monitoring wells and staff gages was conducted using precision optical surveying methods to the nearest 0.1 and 0.01 feet for horizontal and vertical precision, respectively. However, horizontal positions for those requiring moderate precision (e.g., soil borings and storm sewers) were recorded using a hand-held GPS unit with its accuracy to the nearest meter, while elevations for moderate-resolution points were estimated based on ground surface topography.

Prior to soil boring and monitoring well installation in Clearview Landfill, proposed locations were checked for subsurface utilities with the Pennsylvania One Call system. All required street permits were obtained from the City Street Permit Department.

Cross-sectional profiles of stream channels were evaluated by Land and Mapping Services, Inc. in April 2003 using a survey-grade GPS. Determining stream-bottom profiling (including depths) helped assess the erosion, deposition, and flood-related potential in the study area, and for numerical modeling of sediment transport. Figure 3-2 shows 21 measurement locations that were selected, beginning along Cobbs and Darby Creeks upstream of Clearview Landfill, and continuing downstream to approximately the central part of Tinicum Marsh.

### 3.2 Gas Sampling Methods

Extensive soil (landfill) gas sampling was conducted at the site to evaluate the potential of landfill gas migration to the surrounding neighborhood. Gas samples were collected from:

- 127 deep GeoProbe<sup>®</sup> holes (soil borings) drilled throughout Clearview Landfill and surrounding areas (i.e., City Park and Eastwick neighborhood);
- 150 shallow punch holes near residential properties next to City Park;

- 16 stormwater catch basins; and
- 6 permanent dry vapor monitoring wells in City Park.

In addition, modeling of potential vapor migration was conducted to assess potential human exposure to soil (and/or landfill) gas.

### 3.2.1 GeoProbe® Gas Sampling

Gas samples were collected from 127 GeoProbe® borings installed throughout the Clearview Landfill area in February 2003, July 2004, and February 2006. Soil boring installation and sampling were first performed in the City Park and Eastwick neighborhood area in 2003 and 2004, and then in the Clearview Landfill once access to the landfill was obtained in 2006. The depths of soil borings varied from 0.5 to 32 feet below ground surface (bgs), depending on the location. Drilling logs for these borings are provided in Appendix B. Figure 3-3 shows the locations of soil borings and respective sampling years. All gas samples were collected as grab samples using Summa canisters.

Soil borings in the Eastwick neighborhood area were located within street right-of-ways. Prior to sampling, GPS coordinates of each boring location were recorded.

During drilling operations, the Lower Explosive Limit (LEL) was measured for each soil boring hole using standard field monitoring instruments. A meter reading above 25% of the LEL was used as an action level to protect workers.

To collect gas samples, a 1-inch diameter PVC pipe, approximately 2 feet in length, was placed into the 2-inch hole of the GeoProbe® boring with its mouth slightly above ground surface. A rubber stopper or bentonite clay was then placed in the annular space between the pipe and the sides of the hole to create a vapor seal outside the PVC pipe. A copper pipe (¼ inch in diameter and approximately 18 inches in length) was inserted through a rubber stopper plugged in the 1-inch PVC pipe and secured to provide a seal to outside air. The copper tubing was connected to an air pump by flexible tubing. Flexible tubing was then connected from the air pump to a pipe manifold with two valves.

Prior to sample collection, one valve was connected to a Lantec Gem-500 gas meter until all the meter readings such as methane (CH<sub>4</sub>), carbon dioxide (CO<sub>2</sub>), oxygen (O<sub>2</sub>), and percent lower explosive limit (%LEL) had stabilized. After stabilization, a Summa canister was connected to the remaining valve manifold and the valve for the gas meter was closed, while opening the valve to the Summa canister simultaneously. After the filling of the Summa canister was completed, the container was shipped with chain of custody documentation to an EPA contract laboratory program (CLP) laboratory or the USEPA Ft. Meade, MD laboratory for VOC analysis. This gas sample collection procedure was repeated at each GeoProbe® location.

VOCs in gas samples taken in 2003 and 2004 were analyzed using the USEPA Compendium Methods TO-14a, while VOCs in the 2006 gas samples were analyzed by Method TO-15. During the years between 2004 and 2006, Method TO-15, which utilizes high resolution gas chromatography/mass spectrometer (GC/MS), had become the de facto method to analyze gas samples collected with stainless steel evacuated canisters. Method TO-15 offers superior moisture trapping, lower detection limits, and an expanded analyte list to Method TO-14a.

GeoProbe® borings were also monitored in the field using several portable gas analyzers, including the following representative models: a Lantec Gem-500 meter for methane, carbon dioxide, and oxygen; an

Industrial Scientific SP200 LEL meter or equivalent for %LEL; and a MiniRAE 2000 photoionization detector (PID) for total VOCs.

Note that seventeen test pits were proposed originally in the Work Plan to identify the edge of waste disposal areas. However, GeoProbe<sup>®</sup> sampling was determined to be an equally performing, yet less intrusive method with lower potential risks to site workers and residents; therefore, test pits were not performed.

### 3.2.2 Shallow Borehole Gas Sampling

During the first round of gas sample collection from the soil borings installed in City Park in 2003, elevated methane and LEL readings were measured in gas from soil borings near the homes located next to City Park, especially the townhomes in Angelo Place (inset in Figure 3-4). Accordingly, additional gas measurements were used to assess the extent of gas migration to the residential properties.

Prior to sample collection, access agreements were coordinated by the USEPA with property owners. As shown in Figure 3-4, shallow gas holes were installed in April and May 2003 at 150 locations along backyard property lines. Of these, 82 shallow gas measurements were taken around the Angelo Place townhomes due to concerns regarding potential landfill gas intrusion into residences. The townhomes in Angelo Place do not have basements; therefore, the gas samples were collected at foundation level.

At each sampling location, a slam bar was used to install a ½-inch borehole in the subsurface. After reaching a depth of approximately 1.5 to 2 feet bgs, a ½-inch diameter copper tube was placed into the borehole and sealed at ground surface. Clear Viton tubing with a reduced coupling was connected to ¼-inch tubing, which was then connected to the Lantec four-gas meter. The borehole was purged for about 1 minute. After readings of methane, carbon dioxide, oxygen, and balance air had been stabilized, readings for these parameters were recorded. Subsequently, a PID was connected to the tubing and readings for total VOCs were recorded after stabilization. This method of gas sample collection was implemented throughout the sampling locations.

### 3.2.3 Stormwater Catch Basin Sampling

In April 2003, gas samples from 16 stormwater catch basin inlets (denoted with the prefix “CB”) were measured with hand-held instruments for concentrations of methane, oxygen, carbon dioxide, and total VOCs. Sampling locations are shown in Figure 3-4.

### 3.2.4 Soil Vapor Monitoring

Since some of the soil borings installed throughout City Park exerted sufficient gas pressure to cause outgassing from the boreholes, permanent vapor monitoring wells were installed to determine if gas at the perimeter of the landfill near adjacent residences was under pressure.

As a result of these preliminary findings in 2003, six permanent vapor monitoring wells (VM-01 through VM-06) were installed in February 2004 as dry wells (screened above the water table) to allow a permanent soil vapor concentration/pressure monitoring capability. The well locations are shown on Figure 3-4. The vapor monitoring wells were made of 2-inch Schedule 40 PVC and were installed at depths of 5 to 10 ft bgs (see Appendix B for drilling logs). Air pressure gauges were added to each monitoring well to measure the differential of ambient air-to-soil gas pressure over time.

In the case of a landfill, the internal pressure is usually greater than atmospheric pressure. Therefore, under normal conditions, gas generated from a landfill will be released to the atmosphere by both convective (pressure-driven) flow and diffusion. However, for a landfill with no gas collection system (e.g., Clearview Landfill), the extent of the lateral movement of gas can be significant, depending on the characteristics of the cover material and the surrounding soil. As a result, gas is vented in an uncontrolled manner and may accumulate below buildings or in other enclosed spaces near a landfill. Moreover, experience at many sites has shown that under paving, foundations, frozen soil or wet-soil conditions (or where soil is covered by an impermeable media); soil vapor can migrate into structures.

Therefore, landfill gas pressures were measured over time in the area of highest soil LEL/VOCs concentrations (i.e., VM-04) and compared to ambient (normal atmospheric) pressure. The difference between borehole pressure and atmospheric pressure is referred to herein as the ‘differential pressure.’ This study was performed to determine whether gas produced from decaying Clearview Landfill wastes was under pressure or possibly diffusing to the surrounding area.

### 3.3 Surface Water/Sediment Assessment

#### 3.3.1 Prior Investigations

Prior to the RI, several investigations documented contamination in the creeks at the LDCA site and Tinicum Marsh: in September 1983, site investigation collected aqueous and sediment samples from Hermesprot Creek, and upstream background samples (Tetra Tech, 1985); USEPA and FWS conducted a joint site investigation in 1988, collecting surface water and sediment samples from Darby Creek (tidal marsh) and Hermesprot Creek (Garnett Fleming, 1989); surface water and sediment samples were collected in May 1998 from Darby, Cobbs, and Hermesprot Creeks near the potential sources, and background upstream of the sources (Weston, 1999). In general, sediment and surface water samples collected from Darby and Cobbs Creeks near Clearview Landfill indicated that sediment and surface water located near and downstream of Clearview Landfill were contaminated with high concentrations of PAHs, including benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, and benzo(g,h,i)perylene.

Water quality in Darby and Cobbs Creeks upstream of the LDCA site has been the subject of extensive study by the Philadelphia Water Department (PWD) to assess the impacts of point- and non-point sources of contaminated runoff to the watershed. Studies began in the 1970s with the U.S. Geological Survey to quantify the pollutant loads in some of Philadelphia’s streams in order to relate changes in water quality to urbanization. Two of the sampling stations were on Cobbs Creek. One station was on Cobbs Creek at U. S. Route 1 (approximately 8 miles upstream of Clearview Landfill), and the other was on Cobbs Creek at the town of Darby, approximately 4,000 feet upstream of Clearview Landfill. Surface water quality samples were collected at each station monthly and analyzed for conductivity, biochemical oxygen demand (BOD), total phosphate, ammonia, nitrite, nitrate and fecal coliform. The analytical results indicated that, over the ten years of monthly sampling, there was a significant increase in BOD, ammonia, total phosphate and fecal Coliform between Cobbs Creek at Route 1 and downstream at Darby (PWD, February 2000).

In the late 1990s, the PWD undertook a contaminant loading screening assessment for the Darby and Cobbs Creek’s watershed. This assessment included two months of water quality sampling in May and June 1999, and the development of a watershed management model (WMM) for estimating pollutant loads. A table (see “Table 5.1 Comparisons of Load Estimates for Cobbs Creek”) from the report “Draft Technical Memorandum No. 3, A Screening Level Contaminant Loading Assessment for the Darby and Cobbs Creeks Watershed,” (PWD, June 2000), presents these findings. The preliminary results indicated that during the 1999 monitoring period the concentrations of most of the monitored water quality

parameters were lower than during the earlier (1970s – 1980s) monitoring period. The exceptions to this were increases in nitrate and fecal Coliform concentrations. The contaminant loads were attributed to municipal and industrial process water discharges, combined sewer overflows (CSOs), sanitary sewer overflows (SSOs), stormwater and urban drainage, septic tanks discharges and atmospheric sources. CSOs exist near the Clearview Landfill and are a known contributor of sewage to Cobbs Creek during overflow events, in addition to stormwater outfalls both upstream and downstream of Clearview Landfill (PWD, 2000).

### 3.3.2 Watershed Assessment

A watershed is a region in which all the rainfall coming from the land drains to a particular body of water or common point. Watershed assessment is the measurement and use of chemical, physical, and biological properties of water body to determine the current conditions of streams and to improve the health of a watershed if necessary. Besides being part of the National Pollutant Discharge Elimination System (NPDES) permitting process that wastewater treatment plants must comply with in order to operate, watershed assessment is important because it identifies non-point sources of pollution (e.g., Clearview Landfill), predicts their impacts on streams and rivers, and recommends management practices that will help manage watershed health.

The Clearview Landfill is situated mostly within the Darby Creek watershed and partly within the Cobbs Creek watershed (see Figure 2-7). Water in these watersheds is delivered to the Delaware River through the wetlands of the John Heinz NWR.

Initial watershed assessment for Darby Creek was done by a literature search. Water quality and flow characteristics were obtained from the PWD Technical Memorandums (PWD, 2000). Information on CSOs was also obtained from PWD and other regional and local sources.

Moreover, a literature review was also conducted for the Christina River Basin. The Christina River is a tributary of the Delaware River near downtown Wilmington, Delaware. Similarly to Darby Creek, the Christina River Basin is also tidally influenced. The intent was to compare sediment quality and transport in the Christina River Basin to those in the Darby Creek watershed.

In addition to the literature review, surface water, sediment, and leachate seep sampling was performed to evaluate the environmental quality of the watershed. Sediment and contaminant transport were evaluated using the sampling results, flow characteristics of the contributing streams, and tidal flow studies.

Furthermore, based on creek watershed discharge (or flow) data, water volume exchange near 84<sup>th</sup> Street/bridge was assessed to determine approximate flush time for water exiting Darby and Cobbs Creeks near Clearview Landfill only. For this assessment, the area south of 84<sup>th</sup> Street was defined as Tinicum Marsh, whereas the area north of this street was defined as Clearview Landfill. Effort was also made to determine the extent and influence of high tide from Tinicum Marsh.

Any precipitation before and during the sampling events was incorporated into the watershed assessment. During the month prior to the surface water/sediment sampling event from April 15 to May 15, 2002, 3.59 inches of precipitation was recorded at the NOAA weather station located at the Philadelphia International Airport (1 mile east of the Clearview Landfill site). During the sampling event, 1.74 inches of precipitation was recorded. During the month prior to the leachate sampling event between June 9 and July 9, 2002, NOAA recorded 2.56 inches of precipitation. During the sampling event, NOAA recorded 0.51 inches of precipitation.

### 3.3.3 Sediment and Surface Water Sample Collection

In May and June 2002, the first round of sediment and surface water samples (denoted with the prefix “SD/SW”) were collected along Darby and Cobbs Creeks upstream and downstream of Clearview Landfill (Figure 3-5). During this sampling event, 38 surface water and sediment samples were collected at the following locations:

- 10 locations in Darby Creek upstream of Clearview Landfill: SD/SW01 through 10;
- 8 locations in Cobbs Creek upstream of the landfill: SD/SW11 through 18;
- 6 locations adjacent to the landfill: SD/SW19 through 24;
- 6 locations in Darby Creek downstream of the landfill: SD/SW25 through 30;
- 4 locations in marsh: SD/SW31, 34, 35, and 37; and
- 4 locations in the impoundment area within the John Heinz NWR: SD/SW32, 33, 36, and 38.

The impoundment area is connected via a tidal gate to Tinicum Marsh in the John Heinz NWR. In addition, the second round of sediment and surface water sampling from six locations (i.e., SD/SW19, 22, 24, 25, 28, and 30) was performed in October 2005 for the ecological risk assessment.

At each sample location, GPS coordinates were recorded, together with physical parameters measured by a portable Horiba-22 multi-parameter sonde. Field parameters measured included temperature, pH, conductivity, total dissolved solids (TDS), dissolved oxygen (DO), turbidity, salinity, and oxidation-reduction potential (ORP). Field observation (e.g., stream bottom and degree/type of refuse) at each sampling location is described in Appendix C.

Supplemental to the surface water and sediment sampling, a field team conducted qualitative analysis on garbage and streambed composition along the creeks in July 2002. The results are shown in Figures 3-6 and 3-7 for garbage density and streambed composition, respectively. Grain size analysis was also conducted to support streambed characteristics.

#### General Sampling Procedures

A shallow-bottom, inflatable ‘zodiac-type’ boat was used to reach the majority of the sampling locations. Sampling locations SD/SW32, 33, 36, and 38 within the John Heinz NWR impoundment area were reached using a vessel without a motor. Sampling locations SW/SD09 and 10; 19 through 31; and 34, 35, and 37 were reached using a vessel with an outboard motor. The remaining locations were reached from the banks of the creeks or by utilizing chest-waders to walk to the locations, while minimizing disturbance of the stream sediments.

Surface water samples were collected upstream from where samplers walked to each location. In addition, each day samples were collected from downstream to upstream locations, to insure that no cross-contamination of samples was occurring due to sampling activity at a previous location. At each sampling location, surface water samples were collected before sediment samples were collected.

#### Surface Water Sampling

After physical parameter measurements were collected using a multi-parameter sonde, aqueous samples were collected at each sampling location for analysis of the following parameters: Target Compound List (TCL) VOCs; TCL semivolatile organic compounds (SVOCs); Target Analyte List (TAL) total metals; TAL dissolved metals; alkalinity; hardness; total organic carbon (TOC); biochemical oxygen demand



(BOD); chemical oxygen demand (COD); total suspended solid (TSS); total dissolved solid (TDS); total nitrogen; ammonium; nitrate; nitrite; and total phosphorus. TCL VOCs/SVOCs, TCL pesticides/Aroclors, and TAL metals and cyanide analyzed at the CLP laboratory are listed in Tables 3-1 through 3-3, respectively.

Surface waters were also sampled for hexavalent chromium and fecal coliform, but only at selected locations due to short holding times and sampling logistics for these parameters. The samples for these two parameters were collected at SD/SW01, 05, 09, 11, 22 through 30, 37, and SWIMP (impoundment).

### Sediment Sampling

Sediment samples were collected from the same locations as the surface water samples and analyzed for TCL organics, TAL total metals, acid-volatile sulfides (AVS), TOC, and grain size analysis. Of these, AVS forms insoluble metal sulfide, thereby affecting bioavailability and the toxicity of the heavy metals in sediment.

For sampling in the impoundment of the John Heinz NWR, a Ponar sediment sampler was used from the boat to collect sediment samples. The sampling locations in Darby and Cobbs Creeks were accessed, utilizing chest-waders and a floating raft, and large stainless steel, pre-cleaned spoons were used to collect the sediment samples.

#### 3.3.4 Leachate Sample Collection

Seeps occur in the banks of Clearview Landfill along Darby and Cobbs Creeks, and flow into both creeks. These seeps were repeatedly observed during the RI and appeared at several different locations, including creek banks with long faces of seepage at certain times. These seep locations were marked using Global Positioning System (GPS) equipment and are plotted on Figure 3-8. The typical seep included orange-stained water/sediment and had an odor. At the southern end of Clearview Landfill, seeps also had an oily sheen. These seeps were determined to be landfill leachate based on professional experience.

The seep locations were qualitatively observed to be drought-dependent in flow rate (lower flows during drought conditions) and to increase substantially in flow during rainy periods. Many seep areas were ephemeral and many seepage faces were long (several tens of feet long); therefore, the locations plotted on Figure 3-8 must be considered approximate. A few seeps were judged to have sufficient flow to allow for sample collection.

Leachate seep sampling was conducted in July 2002 (first round) and April 2006 (second round). All leachate samples were collected along the Darby/Cobbs Creek banks of the Clearview Landfill. Five leachate seeps (LD-LS/LC01 through 05) were sampled in 2002 and additional five seeps (LD-SEEP2 through 6) were sampled in 2006 at locations depicted in Figure 3-8. All sampling locations were accessed, utilizing chest-waders and a floating raft. These locations had previously been marked and staked during a reconnaissance visit. At each sample location, GPS coordinates were recorded along with physical parameters (e.g., pH, temperature, DO, conductivity, turbidity, salinity, TDS, and ORP) measured by a portable Horiba 22 Multiparameter sonde.

At each sampling location, a pre-slotted PVC pipe was placed into the seeping creek bank. Aqueous samples for each seep were then collected as leachate flowed through the PVC pipe into the sample bottles. Samples were filtered at the site for dissolved parameters using a filtration apparatus attached to a hand-operated pump. In general, all samples were very turbid.

All leachate seeps were analyzed for the following parameters:

- TCL VOCs/SVOCs
- Fecal Coliform
- TSS/TDS
- Nitrite/Nitrate
- TCL Metals (total/dissolved)
- BOD/COD
- TOC
- Total Nitrogen
- Total Phosphorus
- Alkalinity/Hardness
- Ammonium

Leachate seep samples collected at three locations (LD-LS/LC01, 03, and 05) reacted with the hydrochloric acid and sulfuric acid preservatives contained in the sample bottles for VOC, total phosphorus, ammonium nitrogen, total nitrogen, COD, and TOC samples. Bubbling and frothing were evident as the leachate sample was added to the sample bottles. The laboratory notified that preservative acids were neutralized (higher pH than desired) for some of the samples when they arrived at laboratories.

At each sampling location, leachate sediment samples were collected and analyzed for TCL organics and TAL total metals. Note that leachate sediment was considered to be sediment directly exposed to leachate at the seep locations.

### 3.3.5 Tidal Study

To evaluate the tidal conditions in Darby and Cobbs Creeks, six staff gages were installed in January 2003, as shown in Figure 3-9. Each staff gage was surveyed to the nearest 0.01 feet in elevation and 0.1 feet horizontally. Prior to installing staff gages in the proposed locations, their locations were visited to confirm the presence of benchmarks and their suitability. In addition, information on tide water levels at the existing gage located near the confluence of Darby Creek and the Delaware River was obtained from the National Oceanic and Atmospheric Administration (NOAA).

Readings of water levels were taken from approximately 7:30 a.m. until 6:30 p.m. to ensure the recording of an 8½-hour tidal cycle. Measurements were taken at 20-minute intervals at all locations. Once a tidal change was detected, measurements were made at field-determined increments to collect sufficient data for analysis.

To monitor water levels, a three-person field crew was assembled at each staff gage. The first member was responsible for reading Staff Gages 4, 4A, and 5; the second member was positioned to read Staff Gage 3; and the third member was responsible for Staff Gages 1 and 2. Efforts were made to take readings every 20 minutes through one full tidal cycle. Expecting the high tide to come around noon, the crew took readings at 2-5 minute intervals in order to record the most accurate representation of the curve approaching the crest of high tide. At most locations, readings were taken just past the expected high tide time of 4:42 p.m. Data were then compiled and modeled to fit the best representation of the tidal water level fluctuations at each staff gage location.

## 3.4 Soil and Groundwater Assessment

### 3.4.1 Soil Sampling

Soil samples were collected from 127 GeoProbes<sup>®</sup> installed throughout the Clearview Landfill area (see Figure 3-3). Soil sampling was performed via direct push methods using a GeoProbe<sup>®</sup> drill rig unit equipped with a hydraulically powered hammer, which drove a MacroCore<sup>®</sup> tube assembly with a drive shoe containing an inner-liner sampler (approximately 1.5 inches in diameter and 48 inches long). The sampler was driven to depth by the hydraulic hammer connected to drill rods and the drive head of the sample tube. After reaching the target depth, the tool was retrieved, and the sampler containing the subsurface sample was removed. The plastic liner was cut from top to bottom exposing the subsurface

sample. After retrieval of the sample, the core was screened using a photoionization detector (PID) to measure any organic vapors (i.e., VOCs). Following the screening process, a lithologic description of the sample was documented, including observations of moisture, color, staining, and soil type. This process was repeated until refusal or native soil was encountered.

After the completion of sampling, soil samples were organized for laboratory analyses, based on the following characteristics:

1. Observations of high PID readings;
2. Observation of stained or discolored soil, presence of non-aqueous phase liquid (NAPL), etc.; or
3. Coarse grain size stratum if no obvious indication of above contamination was noted.

Typically, three soil samples at different depths were collected from each sampling location, including one surficial sample and two subsurface samples. A surficial soil sample was typically taken within the top six inches of the boring (below asphalt or concrete, if present), while the two subsurface samples were taken at 0.5 to 32 feet bgs. Soil samples at each boring location were analyzed for as many of the following parameters as possible, depending on available sample volumes. Soil samples were analyzed in the following prioritized order:

- TCL SVOCs for surface and two subsurface soil samples;
- TAL Metals for surface soils only;
- TCL VOCs for one subsurface soil per location;
- Pesticides/Herbicides/PCBs for surface soils only;
- TOC with grain size analysis;
- Grain Size for fine particulates using hydrometer;
- TCL organics and TAL metals in transportable soil fraction; and
- Dioxin – 10% of samples if NAPL or ash present.

#### Background Soil Evaluation

In April 2007, background soil samples were collected at a Korman Suites property and the John Heinz NWR. The soil samples were collected from these areas to determine soil concentrations, which could be attributed to natural and/or anthropogenic non-point sources within a local background area, without any contribution from Clearview Landfill.

As Figure 3-10 shows, soil samples were collected from 11 locations using a hand auger, including 4 in the northern part of the John Heinz NWR (PSS-07 through 10) and 7 in the Korman Suites property (PSS-13, 15, 18, 19, 27, 29, and 30). Two samples were collected from each location: a shallow subsurface sample (0 to 6 inches deep) and a deep subsurface sample (12 to 24 inches deep).

The samples were analyzed for VOCs, SVOCs, pesticides, PCBs, and total metals. In addition, soil sample logs for each sampling location were recorded.

#### 3.4.2 Groundwater Sampling

During the RI, extensive groundwater samples were collected from GeoProbe® borings and monitoring wells installed throughout the Clearview Landfill area to collect data and evaluate the nature and extent of contaminated groundwater. Detailed information on sampling activities is provided below.

#### 3.4.2.1 GeoProbe® Borings

Groundwater samples were collected from the GeoProbe® borings when groundwater was encountered (winter/summer 2003; summer 2004; and winter 2005/2006). A 1-inch PVC pipe with a section of slotted screen was installed to the base of the boring. This temporary well allowed groundwater sample collection without collapse of the borehole wall.

Groundwater samples were collected after sealing the annular space around the temporary PVC well and allowing sufficient time for groundwater to enter the well. Dedicated polyethylene tubing (¼-inch diameter) was placed inside the temporary screen to approximately 1 foot below the water level. The tubing was connected to a peristaltic pump, and water was extracted at a low flow rate of approximately 0.5 liters per minute.

Groundwater samples were analyzed for the following analyses:

- TCL organics
- TAL metals and cyanide
- Pesticides/Aroclors
- Dioxin (10% of samples)

Upon completion of groundwater sampling from the soil borings, the temporary PVC pipe was removed from the borehole, and bentonite chips were added into the borehole, followed by water for hydration. For borings installed within the street, the bentonite was compacted into the borehole to approximately 6 to 8 inches bgs. Asphalt patch was then added into the remaining hole and compacted to existing grade. Borings installed in grass-covered areas were abandoned by adding bentonite into the borehole at grade.

#### 3.4.2.2 Monitoring Wells

A total of 15 monitoring wells were installed to determine groundwater quality, flow direction, and gradient (Figure 3-11). Thirteen wells were installed along the landfill perimeter and two wells were constructed in the center of the landfill to map groundwater contours and concentration gradients.

##### Construction

Monitoring well construction was performed by Ameridrill, Inc., using either a GeoProbe® or hollow-stem auger drilling. The monitoring well locations were surveyed through the Pennsylvania One Call system for subsurface utilities prior to drilling. Six wells (MW01S/01D, MW02, MW03, MW08, and MW09) were constructed within City Park in April 2003, and nine additional wells (MW04, MW05S/05D, MW06, MW07S/07D, MW10, MW11, and MW12) were constructed on the landfill in December 2005 through February 2006. Of these, well pairs MW01S/01D, MW05S/05D, and MW07S/07D were installed as couplets to evaluate water quality changes with depths and vertical hydraulic gradient (wells with suffix “S” refer to the shallower well of a well pair, and those with suffix “D” refer to the deeper well.)

During well installation, continuous coring split-spoon sampling to the target depth (or refusal) was typically conducted. A split spoon is the sampler which is assembled by aligning the two halves of the alloy barrel, and then screwing the head piece on top and the drive shoe on the bottom, with a beveled cutting tip. A split spoon is used to take subsurface soil samples by driving it into the soil at the bottom of

a borehole. When the split spoon is pulled to the ground surface, it is disassembled and split in half; therefore, soil samples are retrieved along the entire length of the borehole, or at selected intervals.

Soil samples were analyzed with a PID for VOCs in soil gas, and lithologic descriptions, along with other characteristics such as moisture, staining, fill material, and evidence of trash, were documented for all soil samples (Appendix B).

Monitoring wells were constructed with 2-inch Schedule 40 PVC, except four stainless steel wells MW04, MW5D, MW10, and MW11.

### Well Development

Following completion of construction, depth to water was measured to determine the standing water column at each monitoring well. Well development was initiated to clean out the majority of suspended sediment in the gravel pack and well casing generated during well construction. Development consisted of raising and lowering a submersible pump throughout the water column, and simultaneously pumping out groundwater. This process was repeated until extracted groundwater was relatively free of turbidity (i.e., turbidity is a measure of the degree to which the water loses its transparency due to the presence of suspended particulates.) All extracted purge water was transferred directly to 55-gallon drums that were staged adjacent to the wellhead. After well development was completed, all water collected in drums was labeled for removal and appropriate disposal.

### Sampling

The first round of groundwater sampling from six monitoring wells in City Park (MW01S/01D, MW02, MW03, MW08, and MW09) was conducted in May 2003. The second round of sampling was performed in March 2006 on thirteen wells installed in both City Park and Clearview Landfill (MW01S/01D, MW02, MW03, MW04, MW05S/05D, MW06, MW07S, MW08, MW09, MW10, MW11, and MW12) and additionally in June 2006 on nine wells in Clearview Landfill (MW04, MW05S/05D, MW06, MW07S/07D, MW10, MW11, and MW12).

Low flow sampling methods were utilized during the sampling event. The depth to groundwater was measured in each well prior to purging. Water was purged from each well using a 2-inch diameter submersible pump. Water quality parameters (i.e., ORP, pH, conductivity, and temperature) were monitored with an YSI Model 3560 meter. Groundwater samples were collected after these water quality parameters and the water level in the well had been stabilized, indicating that the water pumped was being drawn from the aquifer. Well sampling logs are provided in Appendix D.

For dissolved metals, groundwater was filtered (in-line) through 0.45-micron acrylic copolymer disposable filters. Dedicated tubing was used at each monitoring well to prevent cross-contamination. Wells that became dry after purging were left to recover overnight and sampled within 24 hours of purging. Groundwater samples were analyzed for the following parameters:

- TCL organics
- TAL metals (total/dissolved) and cyanide
- Pesticides/Aroclors
- Dioxins (10% of samples)
- Anions (e.g., nitrite, nitrate, sulfate, and chloride)

### 3.4.3 Groundwater Flow Model

After groundwater analytical results and hydrologic information were obtained, a site-specific conceptual groundwater model was made utilizing a flow net graphical method and a flow model (MODFLOW®). This graphical method for groundwater flow and level analysis is fully described by Freeze and Cherry (1979).

The equation for Darcy's Law is commonly expressed as:

$$Q = -KA \frac{dh}{dl} = -KA \left( \frac{h_A - h_B}{L} \right)$$

or simply

$$v = Q/A = -K \frac{dh}{dl}$$

Where,

$Q$  = Discharge rate

$A$  = Cross-sectional area

$h$  = Hydraulic head at points A and B

$L$  = Distance between points A and B

$K$  = Proportionality constant (e.g., hydraulic conductivity)

$v$  = Darcy velocity or specific discharge

The term  $dh/dl$  is also referred to as the hydraulic gradient. The negative sign indicates that the flow of water is in the direction of decreasing head. Darcy's law in its simplest form shows that the flow velocity equals the product of the hydraulic conductivity and the hydraulic gradient.

A flow net is generated by mapping the water table elevation as a series of equipotential lines, and then plotting the flow lines perpendicular to the equipotential lines, resulting in generating curvilinear square grids that represent the approximate flow direction (channel). In general, the large square grids indicate locations of increased recharge rate, whereas small grids indicate locations of increased flow. Therefore, the flow net shows a graphical approximation of the flow direction and qualitative flow rates/volume, which in turn generate a "picture" of the aquifer.

The three-dimensional groundwater flow model, Visual MODFLOW® by Waterloo Hydrogeologic, was used to further analyze the hydrologic conditions at the site and estimate the groundwater discharge into the surface water. Visual MODFLOW® is the flow model MODFLOW (originally developed by the USGS) with a graphical user interface. The model estimates hydraulic conductivities, recharge, groundwater levels, and contours of the surface, and all underlying geologic layers. The results of the model were then calibrated using the WinPEST (part of Visual MODFLOW® software package) until the modeling results approximated the field data.

Once the model was adequately calibrated, an additional software package, ZoneBudget, was run to compute subregional water budgets that provide information necessary for estimating the seepage rate of groundwater into the creeks and the Eastwick neighborhood. The mass balance was performed for the

various components, including recharge, evapotranspiration, constant head, river leakage, and flow between the selected zones within the model boundary.

### 3.5 Stormwater Assessment

Stormwater runoff and erosion potential are important issues at Clearview Landfill. Accordingly, drainage assessment, stormwater sampling, and evaluation of erosion potential were performed to assess these characteristics at the landfill.

#### 3.5.1 Drainage Assessment

Based on site topography, building placement, storm inlets, and ground surface elevations, a drainage assessment was conducted to understand on-site sediment and stormwater transport mechanisms, and to determine stormwater sampling locations. As shown in Figure 3-12, seven distinct drainage basins (Basins A through G) were identified at Clearview Landfill and the City Park. In general, stormwater west of the ridge (Basins D, E, and F) drains toward Darby and Cobbs Creeks, while stormwater to the east of the landfill ridge (Basins A, B, C, and G) drains toward the City Park (PD02 and PD03). Drainage from the landfill is also collected in a low lying depression in the southern portion of the City Park (PD01). Ponding in these depression areas was observed during periods of high precipitation.

#### 3.5.2 Stormwater Sampling

The initial stormwater sampling took place in February through May 2003. Outfalls into Cobbs and Darby Creeks near Clearview Landfill were located during a site visit (Figure 3-12). The field team also located ponding areas where stormwater is collected in City Park during a storm event. Samples were collected from eight locations, including five outfalls (CS01-CS05) and three ponding areas (PD01-PD03). Of these, sampling location CS01 was chosen for background.

Three rounds of outfall sampling were conducted, including two during storm events (in April/May 2003) and one during dry weather (May 2003). A storm event was defined as an event when precipitation greater than 0.2 inches was recorded, as per the EPA NPDES Storm Water Sampling Guidance Document (USEPA, 1992). Based on the records at the NOAA weather station located at the Philadelphia International Airport, precipitation during the storm event samplings from outfalls and ponding areas was 0.26 and 0.74 inches, respectively.

The stormwater sampling locations were flagged during an initial site visit in February 2003. Samples were collected using a long-handled, high-density polyethylene dipper. Effort was made not to disturb aqueous samples. If the dipper was mucked during sampling, it was emptied and the sampling location was allowed to settle prior to re-sampling. Samples were filtered with a disposable filter for dissolved matter analyses in the field. All stormwater samples were analyzed for TCL organics and TAL metals (total/dissolved).

#### 3.5.3 Evaluation of Erosion Potential

A screening of erosion potential was performed to assess the possibility of erosion in Clearview Landfill, especially on its side slopes adjacent to banks of Darby and Cobbs Creeks. The screening evaluation included the following tasks:

1. Based on the topographic mapping and stormwater assessment, part of Darby Creek that runs along the Clearview Landfill was assessed.

2. Cross-sectional topographic profiles, including approximate stream-bottom invert elevations, were constructed for the Darby and Cobbs Creeks. Channel invert elevations at representative cross-sections were used to determine the mean channel velocity for estimating the erosion potential. The resulting 21 profiles were constructed for areas with a high erosion potential (see Figure 3-2). These profiles are provided in Appendix E.

### 3.6 Ecological Characterization

#### 3.6.1 Screening-Level Ecological Assessment

A screening-level ecological risk assessment (SLERA) was conducted to evaluate potential risks from contaminants associated with the Clearview Landfill to surrounding ecological receptors. The SLERA represents Steps 1 through 2 in the Ecological Risk Assessment (ERA) process, and was conducted as per the Ecological Risk Assessment Guidance for Superfund (ERAGS): Process for Designing and Conducting Ecological Risk Assessments (USEPA, 1997). Steps 1 and 2 of an ERA (SLERA) are preliminary, initial screening processes designed to estimate the likelihood of ecological risk and to provide a basis for determining the necessity of Step 3 of the ERA. Step 3 of the ERA process involves a refined food web exposure analysis to fully evaluate risk to ecological receptors.

The SLERA evaluated the potential of the site-related contaminants to lower-trophic level organisms, including terrestrial plants and invertebrates, aquatic plants, benthic invertebrates, and fish via a direct toxicity comparison of contaminant concentrations to literature-based toxicity reference values. Upper-trophic level organisms, including terrestrial birds and mammals, as well as aquatic birds, mammals, and reptiles, were evaluated with a food-web analysis that relates measured concentrations to daily dosage due to bioaccumulation from ingestion. The SLERA also determined complete exposure pathways in which plants and animals are exposed to contamination, and identified a number of contaminants of potential concern (COPCs). At the time of the SLERA, USEPA had access to City Park only. Therefore, the terrestrial area evaluated for SLERA was limited to City Park. For detailed information, refer to the SLERA report (Appendix V).

#### 3.6.2 Baseline Ecological Risk Assessment

Additional sampling from the Clearview Landfill site was performed as part of the Baseline Ecological Risk Assessment (BERA), which consists of Steps 3 through 7 of ERA, and concludes with Step 8, a risk management evaluation. The BERA report is included in Appendix V. During the BERA, preliminary direct and food chain COPCs determined during SLERA were further evaluated in order to identify those chemicals that are risk drivers under more realistic exposure scenarios. A focused Ecological Field Sampling Plan (FSP) was submitted to USEPA in July 2005.

Field verification was performed in August 2005 to determine field conditions, stake out sample locations (including background locations) with a GPS, and confirm the abundance/absence of earthworms and aquatic invertebrates. Tidal charts were also obtained and evaluated for determining access to Darby Creek.

In October and November 2005, collection of tissue samples for earthworm exposure was conducted for toxicity bioassays.

For ecological evaluation of surface soil, earthworm tissue samples (*Eisania fetida*) were collected at four locations in City Park (GP021, GP031, GP032, and GP081) and a background location (see Figure 3-13



for locations), and analyzed for inorganics, SVOCs, and pesticide/PCBs. In addition, seven soil samples from City Park and background area (GP021, 031, 032, 035, 037, 081, and W-BACKGROUND) were collected to perform a long-term toxicity exposure study using lab-grown earthworms and Potworms (*Enchytraeus albidus*).

For ecological evaluation of sediment, six sediment samples were collected along Cobbs and Darby Creeks (SD/SW19, SD/SW22, SD/SW24, SD/SW25, SD/SW28, and SD/SW30) and one from a background location (W-BACKGROUND). Sediment bottom-dwelling worms (*Lumbriculus variegates*) supplied by a laboratory were grown in these sediment samples during 28-day bioaccumulation tests, and tissue samples were analyzed for inorganics, SVOCs, and pesticides/PCBs. Seven sediment samples were also submitted for a life-cycle test using lab-supplied freshwater midges.

Three water samples were collected, including two leachate samples from LS/LC02 and LS/LC05 (where the highest concentrations of COPCs were observed during the SLERA), and one surface water sample from SD/SW22 near seepage. These samples were used to conduct 96-hour acute bioassay using the fathead minnow.

All soil, sediment, and surface water samples were analyzed for TAL inorganics, TCL organics, pesticides/Aroclors, grain size (soil/sediment only), and TOC. Major ecological habitats are shown in Figure 3-13, including City Park as Maintained Lawn/Open Field and Clearview Landfill as Disturbed Scrub/Woodland.

### 3.7 Investigation-Derived Waste Characterization and Disposal

An investigation-derived waste (IDW) minimization approach was utilized to address the anticipated IDW generated during the RI. Several types of IDW were generated, including:

- Soil - drill cuttings from well installation and direct push sampling activities;
- Water - produced from equipment decontamination, drilling activities, well development, groundwater sampling, and aquifer testing; and
- Trash – from disposable field clothes and other assorted trash.

The IDW minimization approach for each waste type was as follows:

Soil - In City Park and the Eastwick neighborhood, all soils generated by drilling soil borings or wells were containerized in 55-gallon drums and staged for later characterization and disposal. On Clearview Landfill, drill cuttings were not containerized, and were either used to backfill each borehole or spread on the surface locally in the vicinity of each well or boring location unless gross contamination (e.g., stained soil, presence of NAPL, high percentage refuge, or high monitoring instrument readings) was observed in the field. Drummed waste was eventually disposed of at off-site facility.

Water – Water generated during drilling operations, well development, and equipment decontamination was containerized, characterized, and transported off-site for disposal. All groundwater pumped during sampling was discharged directly to the ground surface, except for visually contaminated groundwater (e.g., oily) that was contained, evaluated, and disposed off-site.

Trash - All disposable field equipment, including sampling implements, disposable field clothes, groves, and other trash and debris, were double-bagged and disposed as municipal solid waste.

Both soil and water samples were analyzed for IDW management purposes. Their analytical results are provided in Appendix F.

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## 4.0 NATURE AND EXTENT OF CONTAMINATION

The purpose of this section is to discuss the nature and extent of contamination at the Clearview Landfill site such that an informed decision can be made in determining the environmental problems associated with the site. Details on sampling results, including specific contaminants detected in various media, their concentrations and locations, are discussed in the following subsections, which follow a similar sequence as Section 3:

- Soil Gas Evaluation
- Sediment and Surface Water Evaluation
- Soil Evaluation
- Groundwater Evaluation
- Stormwater Evaluation
- Ecological Evaluation

### 4.1 Gas Evaluation

Analytical results from multiple sampling events from 2003 to 2006 indicate that contaminants, primarily VOCs and landfill-related gas components, were present in soil gas throughout the Clearview Landfill study area. Gas samples were collected from a total of 127 Geoprobe® borings (GPs), 150 shallow boreholes (SGs), 16 stormwater catch basins (CBs), and 6 permanent vapor monitoring wells (VMs).

#### 4.1.1 Landfill Gas

Landfill gas is produced by the biodegradation of the disposed organic waste, and typically consists of a number of gaseous compounds, including the principal gases (e.g., methane and carbon dioxide), and trace gases (e.g., nitrogen, hydrogen, hydrogen sulfide and VOCs). The composition of landfill gas varies, depending on the actual percentages of the various wastes landfilled and the age of a landfill. Table 4-1 presents landfill-related gases (i.e., CH<sub>4</sub>, CO<sub>2</sub>, CO, O<sub>2</sub>, and H<sub>2</sub>S), %LEL and total VOCs detected in soil gas by hand-held field instruments. Of these, carbon dioxide, %LEL, and methane measured in the soil gas samples from the Geoprobe borings (GPs) and permanent vapor monitoring wells (VMs) are shown on Figure 4-1a; and those from the shallow boreholes (SGs) and stormwater catch basins (CBs) are shown on Figure 4-1b.

In general, methane was detected widely in the soil gas sampling locations on/near Clearview Landfill, but consistently at much lower concentrations (less than 5%, except for several locations) than would be expected from an active or recently closed municipal solid waste (MSW) landfill. No methane was detected in any of catch basins; therefore, it was confirmed that stormwater lines that bisect the landfill do not act as a landfill gas/methane migration conduit.

Active or recently closed MSW landfills that undergo biodegradation (mostly anaerobic) typically have 40 to 60% methane content in landfill gas. The comparatively low methane content in on-site soil gas suggests that (a) most of the organic fraction of landfilled waste has already decomposed; (b) the majority of organics has dissolved and washed out (commonly occurring in a landfill with no engineering cover/liner systems); or (c) organic waste was a relatively small fraction of the original waste content. However, the third assumption is less likely based on historical information (MSW was reportedly a large part of the waste managed at Clearview Landfill.)

As shown in the historical aerial photographs on Figure 1-3, wetlands and a stream were flowing through the site prior to establishment of the landfill between 1953 and 1965. The Geoprobe boring logs collected in the City Park confirmed that there is a 1- to 3-foot thick layer of fill soil over waste materials that filled the former stream channel (see boring logs in Appendix B for subsurface materials encountered). As a result, there is a great deal of moisture mixed with the landfilled waste. With increased moisture content in the waste matrix, the organic material breaks down faster because of accelerated microbial degradation. This has been demonstrated using bioreactor-type landfills with leachate recirculation (Kim and Pohland, 2003). In addition, the hydrogeologic investigation indicated that this former wetland area has high hydraulic conductivity, resulting in rapid discharge of groundwater to Darby Creek.

As shown on Figures 4-1a and b, eleven soil gas sampling locations were detected with methane content (equal to or greater than 5% by volume), all near the residential properties in the City Park, including GP024 (5%); GP029 (13.8%); GP031 (14.6%); GP032 (5%); GP033 (5%); GP035 (5%); GP036 (5%); GP046 (6%); GP055 (5.6%); GP085 (75.5%); and SG024 (58.7%). Of these, methane content at nine locations was within the flammable range (between 5 to 15% by volume).

#### 4.1.2 Soil Gas

Table 4-2 lists 24 VOCs detected in these soil gas sampling locations and their exceedance (if any) of respective EPA Region III Risk-Based Concentrations (RBCs) for ambient air. Figures 4-2, 4-3, and 4-4 show the distributions of total VOCs, benzene (as a substance representative of fuel-related substances), and chloroform (as a substance representative of solvent-related substances) detected in the soil gas samples, respectively. The extent of soil gas contamination in the three investigative subareas is discussed below. Soil gas analytical results are available in Appendix G.

In summary, benzene was the most frequently detected substance in soil gas throughout the Clearview Landfill and the surrounding areas. Within Clearview Landfill, comparatively high levels of VOCs were detected in the central and southern parts of the landfill. More significant levels of VOCs were detected in the northern open field and central parts of City Park. Levels of VOCs detected in the Eastwick neighborhood area were not significant.

##### *Clearview Landfill*

As shown on Figure 4-2, soil gas VOCs generally cluster in the central and southern parts of Clearview Landfill. Fourteen VOCs were detected in soil gas within the landfill at concentrations exceeding the RBC for ambient air, including 1,2,4-trichlorobenzene; 1,2,4-trimethylbenzene; 1,3,5-trimethylbenzene; 1,3-dichlorobenzene; 1,4-dichlorobenzene; benzene; carbon disulfide; chlorobenzene; chloroethane; dichlorodifluoromethane (Freon 12); MTBE; PCE; TCE; and vinyl chloride (VC).

Fuel-related VOCs, such as benzene and MTBE, were detected at a greater frequency in exceedance of their RBCs than solvent VOCs such as TCE. Benzene was detected in most of the samples at concentrations ranging from non-detect to 1,200  $\mu\text{g}/\text{m}^3$ , while TCE was detected only at one location, GP239 at 12  $\mu\text{g}/\text{m}^3$ .

Other VOCs were detected at concentrations ranging from 50 to 2,000  $\mu\text{g}/\text{m}^3$  in the southern portion of Clearview Landfill. The southern portion of Clearview Landfill is currently utilized by several active businesses for industrial/commercial activities, such as automobile salvage, vehicle demolition, drum recycling, truck/auto repair, asphalt/snow removal contractor's equipment storage, and used appliance recycling (see Figure 2-3).

High levels of VOCs (greater than 2,000  $\mu\text{g}/\text{m}^3$  as total) were also detected at the open area in the central part of the landfill (at GP216, 217 and 220). Chlorobenzene and dichlorodifluoromethane were detected most frequently and at high concentrations. Elevated VOCs (3,885  $\mu\text{g}/\text{m}^3$  in total), consisting mostly of benzene and carbon disulfide, were detected at GP230, which was on the western side (near Darby Creek) of the central clearing.

#### *City Park*

Nineteen VOCs were detected in soil gas within the City Park area at concentrations exceeding the RBCs for ambient air, including 1,1,2-trichloroethane; 1,2,4-trimethylbenzene; 1,2-dichloroethane; 1,3,5-trimethylbenzene; 1,4-dichlorobenzene; benzene; benzyl chloride; carbon tetrachloride; chlorobenzene; chloroethane; chloroform; cis-1,2-dichloroethylene; dichlorodifluoromethane (Freon-12); methyl ethyl ketone (2-butanone); methylene chloride; MTBE; PCE; TCE; and VC. Similar to Clearview Landfill, fuel-related VOCs in City Park were detected generally at a greater frequency in exceedance of the respective RBCs than solvent-related VOCs. Benzene was detected at concentrations ranging from non-detect to 680  $\mu\text{g}/\text{m}^3$  (at GP032), while TCE was detected at concentrations ranging from non-detect to 44  $\mu\text{g}/\text{m}^3$ .

Of the sampling locations, elevated concentrations of VOCs (in terms of total VOCs greater than 10,000  $\mu\text{g}/\text{m}^3$ ) were detected in four locations within City Park—two in the northern open field part (12,080  $\mu\text{g}/\text{m}^3$  at GP033 and 12,092  $\mu\text{g}/\text{m}^3$  at VM06) and two in the central part (13,119  $\mu\text{g}/\text{m}^3$  at VM01 and 13,029  $\mu\text{g}/\text{m}^3$  at VM03). High levels of total VOCs from vapor monitoring wells (VMs) consisted mostly of methyl ethyl ketone (2-butanone), which is a common industrial solvent. The elevated level of total VOCs at GP033 was contributed by dichlorodifluoromethane (Freon-12), which was used as a refrigerant and aerosol spray propellant until its manufacture was banned.

#### *Eastwick Neighborhood*

Ten VOCs were detected throughout the Eastwick neighborhood area, including 1,1,2-trichloroethane; 1,2,4-trimethylbenzene; 1,3,5-trimethylbenzene; 1,4-dichlorobenzene; benzene; chloroform; MTBE; PCE; and TCE. Of these, fuel-related VOCs were detected at a greater frequency in exceedance of their respective RBCs than solvent VOCs. In general, levels of VOC detection in soil gas were not significant in this area. Benzene was detected at concentrations ranging from non-detect to 97  $\mu\text{g}/\text{m}^3$ , while TCE was detected at concentrations ranging from non-detect to 3.3  $\mu\text{g}/\text{m}^3$ . The highest level of total VOCs was detected at location GP016, with a concentration of 787  $\mu\text{g}/\text{m}^3$  (mostly chloroform).

#### 4.1.3 Soil Vapor Evaluation

Because of concerns about potential soil gas migration to the nearby residential properties, soil vapor pressure was analyzed. The USEPA also performed indoor air intrusion modeling utilizing the soil gas data. The findings from these analyses are summarized below.

##### *Soil Vapor Pressure Analysis*

The differential of ambient air-to-soil gas pressure was measured at six VMs installed in City Park. The pressure reading logs for these vapor wells are contained in Appendix H. Of these wells, VM-04 (installed near the backyards of the Angelo Place townhomes) contained the highest concentration of VOCs. Table 4-3 lists the soil pressures measured in April through May 2004 at VM04 and the calculated pressure differentials. These results are graphically presented on Figure 4-5.

The results show that there is not a great deal of landfill gas pressure within the landfill. As the atmospheric pressure rose, the landfill gas pressure dropped. Conversely, as the atmospheric pressure dropped, the landfill gas pressure rose. At times, measured soil vapor pressures at VM04 were higher than atmospheric pressure. Higher soil vapor pressures in subsurface could cause partial venting of some soil gases into residences near Clearview Landfill under certain conditions (such as wet or covered wastes).

Residents noted odors during a public meeting; however, no complaints of odors from residents were reported by the City of Philadelphia Fire Department, and none were reported by Korman Suites (the current owner of most of the Angelo Place townhouses). Furthermore, no visible cracking of foundations/walls was observed during geotechnical inspections (outside only) of the Angelo Place homes in 2003/2004. To determine whether any soil gas venting into homes was actually occurring, USEPA performed a screening air analysis.

#### *Vapor Intrusion Analysis*

A screening indoor air analysis was performed by USEPA using July 2003, July 2004, and January/February 2006 soil gas data from six VMs and 40 Geoprobe borings installed in the City Park and Eastwick neighborhood. The purpose of this preliminary analysis was to determine whether vapor intrusion of contaminants in the soil gas into the residential properties in Eastwick neighborhood may be of concern. The screening version of the Johnson & Ettinger soil gas-to-indoor air model (Version 2.0) was used. The full reports are available in Appendix I, including vapor intrusion analyses on soil gas samples collected in 2003, 2004, and 2006.

The soil gas data used in this analysis were taken from the entire open borehole, above the saturated zone at an average depth of 15 feet bgs. Soil gas samples were also taken at six VMs (VM01 through VM06) near the landfill border. These dry vapor well samples were drawn at a depth of 9 feet. Since the field blanks were contaminated with low concentrations of mainly BTEX compounds and refrigerants, some of the data in the samples were not used for this analysis. In addition, pairs of duplicate samples collected during sampling, including VM06/VM07, GP89/GP113, GP91/GP111, and GP109/GP112, were included in the screening. The maximum contaminant concentrations found between the duplicate pairs were used for analysis.

The following compounds found in the soil gas samples were not listed in the Johnson & Ettinger chemical database; subsequently, they were not modeled.

- 4-Ethyltoluene in samples collected from VM01, VM05, GP17, GP44, GP81, GP83, GP84, GP86, GP87, and GP88;
- 1,2-Dichloro-1,1,2,2-tetrafluoroethane in samples from VM02, GP10, GP83, GP84, GP85 and GP86; and,
- 2-Hexanone in sample from GP84.

Xylene was listed as m-, p- and o-xylene in the soil gas data. Of these, m,p-xylene was modeled with the combined concentration existed as all m-xylene, while o-xylene was modeled separately.

The parameters used with the Johnson & Ettinger model included:

Model Parameter	Input
Soil Conservation Service soil classification	Silt loam (derived from Geoprobe logs at each sampling location); Default silt loam soil properties
Depth below ground surface	Sample specific, but average of 15 feet; vent wells modeled at 6 feet
House structure and dimensions	Default two-story home with basement
Air exchange rate	Default - 0.25/hr
Q soil	Default - 5 L/min

The results from the screening indoor air analysis are listed in Table 4-4. The only compounds that exceeded their respective ambient air RBC concentrations were benzene at one location, GP98, and chloroform at one location, GP16. For benzene, the calculated indoor air value was 4.24E-01  $\mu\text{g}/\text{m}^3$ , and the RBC value was 2.30E-01  $\mu\text{g}/\text{m}^3$ . For chloroform, the calculated indoor air value was 3.32E-01  $\mu\text{g}/\text{m}^3$  and the RBC value was 7.7E-02  $\mu\text{g}/\text{m}^3$ .

#### 4.2 Sediment and Surface Water Evaluation

This section discusses the findings from sediment and surface water sampling events, including both the 2002 first round of sampling at 38 locations, and the 2005 second round of sampling at six of the locations sampled in 2002. Leachate sediment and aqueous samples collected from 10 seep locations at Clearview Landfill in 2002 and 2006 are also discussed in this section. Descriptions of field observations are provided in Appendix C.

Because of the urban watersheds feeding Darby and Cobbs Creeks, major upstream contaminant sources (see Section 3.3.1) are potential sources of contamination in Darby and Cobbs Creeks. For this reason, 18 background sediment and surface water samples were collected. The majority of these samples showed significant contamination.

Table 4-5 presents the results of sediment grain-size analyses. Tables 4-6a through d and 4-7a through d show full lists of substances detected in the sediment and surface water samples, respectively. Table 4-8 lists concentrations of water quality parameters in surface water. Table 4-9 presents the relationship between PAH contamination and sediment grain size.

##### 4.2.1 Sediment Sample Analytical Results

Of the contaminants detected in the sediment samples, PAHs were detected more frequently and at much higher concentrations than any other class of contaminants. VOCs and pesticides were detected in relatively low concentrations in both the 2002 and 2005 sediment samples. The complete analytical results are provided in Appendices J and K for stream sediment and leachate sediment, respectively.

Figure 4-6 shows concentrations of selected PAHs in sediment and potential contributors of contamination (i.e., Folcroft Landfill, Tank Farm, and Clearview Landfill) in relative distance. This figure clearly shows that sediment in Darby Creek near these sources contained higher PAH concentrations than

sediment upstream and downstream of these sources. Moreover, the second plot on Figure 4-6 shows that sediment in Cobbs Creek upstream of Clearview Landfill was also significantly contaminated with PAHs.

Figure 4-7 shows the spatial distribution of total PAHs in sediment. PAHs were highest at location SD24 immediately downstream of Clearview Landfill. In addition, immediately upstream of SD24, a visible oily sheen with an associated petroleum odor was repeatedly noticed during sampling. The sheen seemed to be seeping from the east bank of Darby Creek (located at the creek bank nearest the southern industrial area on Clearview Landfill). Monitoring well MW04 also encountered free-product oil when it was drilled within 200 feet of this oily seepage area (see Figure 3-11 for location). However, Figure 4-7 shows that total sediment PAH concentrations at SD24 in 2002 significantly decreased in 2005 (from 120,170 to 2,640  $\mu\text{g/kg}$ ). Similar trends were also observed for downstream locations SD25, 28, and 30.

The screening values used to evaluate the extent of sediment contamination were the USEPA Region 3 RBCs for residential soil (October, 2007) and the Biological Technical Assistance Group (BTAG) Screening Benchmarks for freshwater sediment (October, 2006). Note that the RBCs were multiplied by a factor of 10 to account for less exposure frequency for recreational contact with sediment (i.e., 26 days per year) compared to residential soil contact (350 days per year). In addition, RBCs were also multiplied by 0.1 for noncarcinogenic substances to create a safety factor. The use of these modified RBCs is consistent with the screening process during human health risk assessment (HHRA), as per USEPA Region 3 guidance. The BTAG Screening Benchmarks are referred to as Ecological Screening Values (ESVs) herein.

Concentrations of benzo(a)anthracene, benzo(a)pyrene, and dibenz(a,h)anthracene detected in sediment samples collected in the study area are shown on Figure 4-8 through 4-10, respectively. Five sampling locations SD23 through 27 in Darby Creek adjacent to/downstream of the landfill exhibited elevated concentrations of benzo(a)anthracene (above its RBC of 2,200  $\mu\text{g/kg}$  and ESV of 108  $\mu\text{g/kg}$ ), ranging from 3,300  $\mu\text{g/kg}$  to 10,000  $\mu\text{g/kg}$ . The highest detection was observed at SD24, located immediately downstream of Clearview Landfill. Benzo(a)pyrene concentrations in sediment exceeded its RBC and ESV of 220  $\mu\text{g/kg}$  and 220  $\mu\text{g/kg}$ , respectively, in all samples; the highest concentration (6,300  $\mu\text{g/kg}$ ) was also observed in the SD24 sample. All sediment samples exhibited dibenz(a,h)anthracene concentrations above the screening value of 220  $\mu\text{g/kg}$  (RBC) or 33  $\mu\text{g/kg}$  (ESV).

Arsenic was detected above its RBC of 4.3 mg/kg or ESV of 9.8 mg/kg at four sampling locations downstream of the landfill (Figure 4-11): SD21 (5.6 mg/kg), SD23 (4.8 mg/kg), SD24 (5.3 mg/kg), SD25 (5.9 mg/kg), and SD27 (4.6 mg/kg). Other metals that were detected above their respective screening values in sediment samples collected from Darby Creek near Clearview Landfill were antimony, cadmium, copper, iron, manganese, mercury, nickel, silver, and zinc.

In summary, the following contaminants were detected at concentrations exceeding the screening values, in at least one sediment sample:

#### PAHs

- Acenaphthene
- Benzo(a)anthracene
- Benzo(b)fluoranthene
- Chrysene
- Fluoranthene
- Indeno(1,2,3-C,D)pyrene
- Pyrene
- Anthracene
- Benzo(a)pyrene
- Benzo(k)fluoranthene
- Dibenz(a,h)anthracene
- Fluorene
- Naphthalene



#### SVOCs

- 2,4-Dimethylphenol
- 4-Methylphenol (p-cresol)
- Dibenzofuran
- 2-Methylnaphthalene
- Bis(2-ethylhexyl)phthalate

#### Pesticides

- Dieldrin
- Heptachlor Epoxide
- DDD
- DDT
- Endrin
- Methoxychlor
- DDE

#### TAL Metals

- Antimony
- Cadmium
- Iron
- Mercury
- Silver
- Arsenic
- Copper
- Manganese
- Nickel
- Zinc

#### VOCs

- Carbon Disulfide
- Chlorobenzene

##### 4.2.1.1 Leachate Seep Sediment

Leachate sediment samples were collected at 10 leachate seep locations, LS/LC01 through LS/LC05 in July 2002 and at locations SEEP2 through SEEP6 in April 2006. Their analytical results are presented in Tables 4-6b and d. In general, the seep sediments had elevated concentrations of total metals and PAHs, and relatively low concentrations of SVOCs, pesticides, and VOCs.

Total PAH concentrations in the leachate sediment samples are shown on Figure 4-12. As with stream sediment samples, PAHs were the most frequently detected type of organic compounds in all leachate sediment samples, and their concentrations were greater than those of any other class of organic compounds. Many PAHs in the leachate sediment samples were detected over their respective screening values. The screening values (either RBCs or ESVs) were exceeded for acenaphthene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, dibenz(a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3-C,D)pyrene, and pyrene. Of these, benzo(a)anthracene, benzo(a)pyrene, and dibenz(a,h)anthracene concentrations are shown on Figures 4-13 through 15, respectively.

The leachate sediment samples at the leachate seep locations also contained elevated levels of metals, especially aluminum and iron, exhibiting concentrations between 7,930 to 23,300 mg/kg and 17,100 to 31,800 mg/kg, respectively. However, aluminum concentrations were below its RBC of 78,000 mg/kg (no ESV for aluminum). Arsenic was detected in exceedance of the screening values (4.3 mg/kg of RBC or 9.8 mg/kg of ESV) in five of the ten sediment samples collected at the seep locations. Other metals that were detected above their respective screening values in leachate sediment samples were cadmium, copper, iron, manganese, mercury, selenium, and zinc.

Pesticides (i.e., dieldrin, endrin, heptachlor epoxide, DDD, DDE, and DDT) and one VOC (i.e., chlorobenzene) were detected above their screening values in several seep sediment samples.

#### 4.2.1.2 Tinicum Marsh Sediment

Sediment samples from locations SD28 through SD38 were considered sediments at Tinicum Marsh in the John Heinz NWR. As with the leachate seep and stream sediment samples, the sediment samples from Tinicum Marsh contained high concentrations of metals and PAHs, and relatively low concentrations of pesticides and VOCs.

As shown on Figures 4-6 through 4-10, concentrations of PAHs in sediment at Tinicum Marsh (except SD31) were generally lower than upstream of the marsh as measured in Darby and Cobbs Creeks. Twelve PAHs were detected above their respective screening values in Tinicum Marsh sediment, including acenaphthene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, dibenz(a,h)anthracene, fluoranthene, indeno(1,2,3-C,D)pyrene, and pyrene. These PAHs were detected at all sediment sampling locations within Tinicum Marsh, but more significant concentrations were observed at locations near Folcroft Landfill and Annex (e.g., SD28, SD29, SD30, and SD31).

Stream sediment at Tinicum Marsh and the impoundment in John Heinz NWR appeared to have higher arsenic concentrations than sediment in the upstream area. Total arsenic was detected at concentrations above the screening value in all sampling locations, ranging from 6.5 to 15.1 mg/kg. The highest arsenic concentration was detected at SD35, which was the farthest downstream location.

#### 4.2.1.3 Background Sediment Quality

Samples collected from SD01 through SD10 represent the background sediment quality in Darby Creek upstream of Clearview Landfill. Samples from SD11 through SD18 represent background sediment quality in Cobbs Creek upstream of the confluence with Darby Creek.

The upstream sediments in Darby and Cobbs Creeks had elevated concentrations of metals and PAHs, and relatively low concentrations of pesticides and VOCs. As shown on Figures 4-6 and 4-7, organic concentrations immediately downstream of Clearview Landfill were higher than those upstream. In addition, the total concentration of PAHs in background sediment was higher in Cobbs Creek upstream of the landfill than in Darby Creek upstream of the landfill.

The widespread elevated background concentrations of PAHs, other organic compounds, and metals are expected, given that the watershed is highly urbanized and receives discharges from industrial and municipal processes, combined sewer overflows, sanitary sewer overflows and stormwater and urban non-point drainage. This also confirms the findings of earlier sediment sampling and is consistent with literature describing urban watersheds (PWD, 2000).

Contamination in Darby Creek sediment downstream of the landfill contained substances from nearby/upgradient sources, including Clearview Landfill (particularly PAHs, pesticides, and metals). Therefore, the source of sediment contamination downstream of the landfill may be related to Clearview Landfill, but it is also likely that a significant portion can be attributed to other sources in the area, as described previously.

#### 4.2.2 Surface Water Sample Analytical Results

In general, the surface water samples contained far fewer analytes at much lower concentrations than the sediment samples. Tables 4-7a through 4-7d show the analytes detected in the surface water samples.

Appendix L provides a complete set of analytical results. For discussion purposes, analytical results were compared to the modified RBCs for tap water (multiplied by a factor of 10 in accordance with USEPA Region 3 guidance, since surface water in the study area is not likely used as a drinking water source) and ESVs (i.e., October 2006 BTAG Screening Benchmarks for freshwater).

Only two pesticides, including dieldrin and heptachlor, were detected in surface water samples at concentrations exceeding their respective RBCs (0.042 and 0.149 µg/L, respectively) or ESVs (0.056 and 0.0019 µg/L, respectively). Dieldrin was detected only at one location SW11 (0.051 µg/L), while heptachlor was detected at two locations in SW15 (0.02 µg/L) and SW18 (0.032 µg/L), all located in Cobbs Creek upstream of the landfill.

No VOCs and SVOCs were detected at elevated levels in surface water samples. Low levels of PAHs, including anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(a)fluoranthene, fluoranthene, indeno(1,2,3-C,D)pyrene, and pyrene, were detected above or equal to their screening values at four locations (SW11, SW13, SW16, and SW19) in Cobbs Creek upstream of the landfill; one location (SW7) in Darby Creek upstream of the landfill; and one location (SW28) in Darby Creek downstream of the landfill. Of these locations, two locations (SW11 and SW13) in Cobbs Creek upstream of the landfill were more frequently detected with these PAHs than other locations. This indicates that Clearview Landfill does not have a significant impact on surface water insofar as PAHs are concerned.

Several VOCs were detected at low concentrations, but above their screening values, including carbon disulfide; TCE; 1,1,2,2-tetrachloroethane; PCE; and toluene. Carbon disulfide was detected above its ESV (0.92 µg/L) at only one location (SW16 [2 µg/L]) in Cobbs Creek upstream of the landfill. TCE concentrations above its screening values (0.264 and 21 µg/L for RBC and ESV, respectively) were detected at four locations—SW29 (0.5 µg/L) and SW34 (0.6 µg/L) located at Tinicum Marsh, and SW17 (2 µg/L) and SW19 (0.7 µg/L) in Cobbs Creek upstream of the landfill. 1,1,2,2-Tetrachloroethane was detected above its screening values (0.53 and 610 µg/L for RBC and ESV, respectively) at SW22 (5 µg/L), SW24 (1 µg/L), and SW26 (3 µg/L), which were located adjacent to the landfill or just downstream of it. One location (SW17) in Cobbs Creek north of the landfill exhibited PCE (2 µg/L) above its RBC value (1 µg/L), but below its ESV value (111 µg/L). Three locations in Cobbs Creek upstream of the landfill were detected with toluene above its ESV (2 µg/L), but below RBC (22,711 µg/L)—SW11 (6 µg/L), SW13 (9 µg/L), and SW14 (5 µg/L). Surface water in Darby Creek upstream of the landfill (SW01 through SW10) did not exhibit any VOCs and SVOCs above screening values.

Many inorganics in surface water samples were detected above their respective RBCs or ESVs. Of all inorganics detected, total and dissolved arsenic concentrations are shown on Figures 4-16 and 4-17, respectively. All sampling locations with arsenic detection exhibited total and dissolved arsenic concentrations greater than its RBC (0.45 µg/L). Total arsenic concentrations above ESV (5 µg/L) were detected at four locations: SW32 (6.9 µg/L), SW33 (9 µg/L), SW36 (7.7 µg/L), and SW38 (9.4 µg/L) within the impoundment at John Heinz NWR; and one location SW11 (7.2 µg/L) in Cobbs Creek upstream of Clearview Landfill. However, no samples were detected with dissolved arsenic above its ESV value.

In summary, the following contaminants were detected at concentrations exceeding the screening values, in at least one surface water sample:

#### PAHs

- Benzo(a)anthracene
- Benzo(b)fluoranthene
- Anthracene
- Pyrene
- Benzo(a)pyrene
- Indeno(1,2,3-CD)pyrene
- Fluoranthene

#### Pesticides

- Dieldrin
- Heptachlor

#### VOCs

- Carbon Disulfide
- 1,1,2,2-tetrachloroethane
- Toluene
- PCE
- TCE

#### TAL Metals

- Aluminum
- Barium
- Cadmium
- Manganese
- Arsenic
- Beryllium
- Iron
- Thallium

#### 4.2.2.1 Water Quality Parameters

General water quality parameters were also assessed for surface water samples. The results are summarized in Table 4-8.

##### *Alkalinity*

Total alkalinity in the surface water samples ranged from 37.2 to 93.8 mg/L as CaCO<sub>3</sub>, which is in a typical range of fresh water levels of 20-200 mg/L. Alkalinity refers to how well a water body can neutralize acids. Alkalinity measures the amount of alkaline compounds in water, such as carbonates (CO<sub>3</sub><sup>2-</sup>), bicarbonates (HCO<sub>3</sub><sup>-</sup>), and hydroxides (OH<sup>-</sup>). These compounds are natural buffers that can remove excess hydrogen ions (which lower pH) that have been added from sources such as acid rain or leachate from Clearview Landfill. Alkalinity also mitigates or relieves metal toxicity by using available buffers to take metals out of solution (i.e., precipitation), thus making it unavailable to aquatic life. Total alkalinity levels below 10 mg/L indicate that the water is poorly buffered, and is very susceptible to changes in pH from natural and human-caused sources.

##### *Oxygen Demand*

Figures 4-18 and 4-19 show the 5-day biochemical oxygen demand (BOD<sub>5</sub>) and chemical oxygen demand (COD) measured in the surface water samples. The surface water samples from two upstream locations SW14 and 15 in Cobbs Creek were detected with BOD<sub>5</sub> greater than the PA Water Quality standard of 10 mg/L. Surface water in Darby Creek, both upstream and downstream of Clearview Landfill, was relatively low in BOD<sub>5</sub>. However, four downstream locations (SW32, 33, 36, and 38) within the impoundment at John Heinz NWR were detected with BOD<sub>5</sub> greater than the PA Water Quality standard. Similarly, high COD values were also detected in the surface water samples collected in Cobbs Creek upstream of Clearview Landfill and the impoundment downstream of the landfill.

Anything that can be oxidized in the receiving water with the consumption of dissolved molecular oxygen is termed “oxygen-demanding material.” This material usually includes biodegradable organic matter, but also includes certain inorganic compounds. The consumption of dissolved oxygen (DO) poses a threat to fish and other higher forms of aquatic life. Any discharges from point and non-point sources (e.g., Clearview Landfill and others upstream of the landfill) that enter the water contribute to the depletion of DO. Organic oxygen-demanding materials are commonly measured by determining the amount of oxygen consumed during degradation in a manner approximating degradation in natural waters.

If oxidation of an organic compound is carried out by microorganisms using the organic matter as a food source, the oxygen consumed is known as BOD and commonly tested over a 5-day period (BOD<sub>5</sub>). The greater amount of organic matter present, the greater amount of oxygen utilized. In contrast, the COD is a measured quantity of organic matter that does not depend on microorganisms. In the COD test, a strong chemical oxidizing agent (chromic acid) is mixed with a water sample, and then boiled. The difference between the amount of oxidizing agent at the beginning of the test and that remaining at the end of the test is used to calculate the COD. The COD value is typically higher than the BOD value because it measures oxygen demand to degrade organic matter both chemically and biologically.

#### *Fecal Coliform*

As Figure 4-20 shows, fecal coliform bacteria were common in surface water samples. Concentrations of fecal coliform greater than 200/100 mL, which is a PA Water Quality standard and a Delaware River Basin Commission regulation, were found in three upstream samples: SW01 and SW05 in Darby Creek, and SW11. SW11 is the furthest upstream location of the Clearview Landfill in Cobbs Creek and exhibited the highest fecal coliform concentration (6,400/100 mL) of all surface water samples. Fecal coliform was also significant in seven of nine downstream surface water samples. Of these, the surface water sample at SW37 contained the highest count of fecal coliform (1,100/100 mL).

Microorganisms found in wastewater (or other contaminated water) include bacteria, viruses, and protozoa excreted by diseased persons or animals. When discharged into surface waters, they make the water unfit for drinking. If the concentration of pathogens is sufficiently high, the water may also be unsafe for swimming and fishing. Fecal coliform is present in the feces and intestinal tracts of humans and other warm-blooded animals, and can enter water bodies from human and animal waste. If a large number of fecal coliform (over 200 colonies/100 mL of water sample) are found in water, it is assumed that pathogenic organisms are also present in the water.

#### 4.2.2.2 Aqueous Leachate Seep

Five aqueous leachate samples were collected in July 2002 (samples LS/LC01 through LS/LC05) and five more in April 2006 (SEEP2 through SEEP6), from the banks of Clearview Landfill along the Darby and Cobbs Creeks. The sampling locations are shown on Figure 3-8. The analytical results are listed in Tables 4-7b and 4-7d (from Appendix K).

In general, the analytical results for aqueous leachate samples indicated that organic compounds were much less common in leachate samples than were inorganic substances. Similar to the findings in the surface water samples, the aqueous leachate samples did not contain many organic compounds such as SVOCs, PAHs, VOCs, and pesticides. Only metals were detected at elevated concentrations, and generally at higher levels than the surface water samples.

Of the pesticides analyzed from the aqueous leachate samples, beta BHC, heptachlor, and heptachlor epoxide were detected at concentrations above their respective RBC values (0.37, 0.15, and 0.074 µg/L).

or ESV values (not available, 0.0019, and 0.0019 µg/L). Distribution of beta BHC is shown on Figure 4-21. Beta BHC was detected at all aqueous leachate sample locations, but only two locations, LS/LC-3 (0.51 µg/L) and LS/LC-4 (0.43 µg/L), exhibited concentrations exceeding the screening value (RBC in this case). Beta BHC levels in aqueous leachate samples collected in 2006 were lower than levels from the 2002 sampling event. Heptachlor was detected at four seep locations—SEEP2 (0.049 µg/L), SEEP3 (0.052 µg/L), SEEP4 (0.036 µg/L), and SEEP5 (0.065 µg/L). Heptachlor epoxide was detected at SEEP4 (0.012 µg/L) only.

The aqueous leachate samples contained elevated concentrations (above their respective screening values) of many inorganics, including aluminum, arsenic, barium, beryllium, cadmium, copper, iron, manganese, mercury, nickel, selenium, silver, thallium, vanadium, and zinc. Therefore, many metals found in the surface water appeared to be leachate-related. Of these metals, total and dissolved arsenic concentrations are shown on Figures 4-22 and 23, respectively. There was no arsenic detected in the dissolved form in the 2002 samples (LS/LCs); therefore, most arsenic was associated with (adsorbed onto) suspended particles in leachate. In contrast, more arsenic was detected in the dissolved form than the suspended form for the 2006 leachate samples (SEEPs). All seep sampling locations detected with arsenic had arsenic concentrations above its screening value (0.45 or 5 µg/L for RBC and ESV, respectively), and values were also significantly higher than those observed in the surface water samples.

Figures 4-24 and 25 show total and dissolved iron concentrations in the aqueous leachate samples, respectively. Iron can exist as soluble ferrous iron ( $\text{Fe}^{2+}$ ) or as the relatively insoluble ferric form ( $\text{Fe}^{3+}$ ) in water. In the 2002 samples, all five sampling locations contained high concentrations of total iron, but much less concentrations (<1,000 mg/L) in the dissolved form (likely ferrous form). When soluble ferrous iron is exposed to oxygen, it oxidizes to the relatively insoluble iron (i.e., suspended colloidal and particulate iron) that is responsible for discolored water. However, the 2006 samples contained more dissolved iron than insoluble iron.

Water quality parameters were also analyzed for the aqueous leachate samples collected in 2002. The BOD<sub>5</sub> results ranged from 32.1 mg/L to 378 mg/L, while COD results ranged from 75 mg/L to 1,800 mg/L (see Figures 4-18 and 19, respectively). All of the parameters are fairly low for landfill leachate, indicating that the seeps consist of large amounts of groundwater mixed with the leachate.

Fecal coliform bacteria were high in the leachate samples (Figure 4-20), especially at two locations—LS/LC01 and 05 (6,000/100 mL). For comparison, the surface water sample taken from SW11 on Cobbs Creek upstream of the landfill also contained a high number of fecal coliform (6,400/100 mL).

#### 4.2.2.4 Tinicum Marsh Surface Water

Samples SW32 through SW38 were collected at Tinicum Marsh in the John Heinz NWR. Surface water at Tinicum Marsh had relatively low concentrations of a few SVOCs and VOCs, and no PCBs were detected. Generally, there were fewer organic compounds at lower concentrations in Tinicum Marsh, compared to upstream surface water samples.

Eight inorganics, including aluminum, arsenic, barium, beryllium, cadmium, iron, manganese, and thallium, were detected at concentrations above their respective RBCs or ESVs. Note that ESVs for inorganics are several orders of magnitude less than RBCs, except for arsenic. Of these, aluminum, barium, iron, and manganese were frequently detected above their respective screening values throughout Tinicum Marsh. Water quality parameters such as fecal coliform, BOD<sub>5</sub>, and COD indicated that this area was impacted by upstream sources, other sources, or both.

#### 4.2.2.5 Background Surface Water Quality

Analytical results for the samples collected at SW01 through 10 present the background surface water quality in the Darby Creek upstream of Clearview Landfill (see Figure 3-5 for sample locations), and those at SW11 through 18 represent background water quality in Cobbs Creek upstream of its confluence with Darby Creek. The analytical results generally indicated that surface water samples in Darby Creek upstream of the landfill contained fewer analytes and at lower concentrations than surface water in Cobbs Creek or Darby Creek downstream of the landfill. In addition, surface water samples in Cobbs Creek upstream of the landfill had high concentrations of metals (see Figures 4-16 and 17), and other water quality parameters such as BOD, COD, and fecal coliform (see Figures 4-18, 19, and 20, respectively).

The high background levels in Cobbs Creek surface water upstream of Clearview Landfill are not surprising given the urbanization in the area, known surface water contamination reported by Philadelphia Water Department (PWD, 2000), and the numerous stormwater/sanitary sewer outfalls into Cobbs Creek that receive discharges from industrial and municipal processes, combined sewer overflows, sanitary sewer overflow, stormwater non-point runoff, and urban drainage.

#### 4.2.3 Sediment Transport Evaluation

Many processes affect contaminant concentrations in Darby and Cobbs Creeks. Several of the more important factors were evaluated during the RI:

- Background contaminant concentrations in this urban area
- Sediment grain size
- Tidal influences

Many substances that were identified as being site-related were detected at far higher concentrations in creek sediment samples than in surface water samples. PAH compounds such as benzo(a)pyrene are found widely in Darby Creek sediment, seeps from the landfill, and also in Cobbs Creek sediments upstream from the landfill. High PAH concentrations have significant impact on the ecosystem since bottom-dwelling organisms are an important food supply for birds, fish, mammals, and larger organisms. The major concern is that some PAHs form carcinogenically-active metabolites (benzo[a]pyrene is the prime example), and PAH concentrations in sediments have been linked with liver neoplasms (an abnormal new mass of tissue that serves no purpose) and other abnormalities in bottom-dwelling fish. Elevated PAH concentrations may therefore pose a threat to aquatic organisms and potentially also to human consumers of fish and shellfish.

Partitioning (the natural processes that can result in contaminants preferentially adsorbing on sediment are collectively called “partitioning”) results from many chemical and physical affinities expressed by compounds. In essence, tar-like PAH compounds are strongly attached to sediment.

Sediment naturally moves with currents and tides. The mobile fraction of the sediment moves rapidly in Darby and Cobbs Creeks. Additionally, contaminants may adsorb more strongly on certain grain sizes. This effect may cause some contaminants to be carried farther than others, as they move with the sediment in different sizes to which they are most strongly adsorbed.

To evaluate the effects of contaminant migration with sediment, a grain size measurement analysis for sediment samples was performed in conjunction with a quantitative analysis approximating sediment movement as it relates to the measured distribution of contamination and grain size.

#### 4.2.3.1 Sediment Grain Size Evaluation

Table 4-5 shows the grain sizes of the sediment samples collected from the stream bottoms of Darby and Cobbs Creeks in May and June 2002 (see Appendix M for full analysis). These sediment samples were classified by grain sizes according to the Udden-Wentworth scale (Wentworth, 1922).

Based on this scale, sediments in the creeks were mostly sand or silt in different sizes. Specifically, sediment in Darby Creek upstream of the landfill consisted of fine and very fine sand. Sediment in Cobbs Creek upstream of the landfill consisted of silt (SD12 and 13), medium/fine sand, and very coarse sand (SD18). In the sediment of Darby and Cobbs Creek adjacent to the landfill, fine/very fine/coarse sand and silt were predominant. This section of the stream bottom in Darby and Cobbs Creeks was covered by extensive waste or refuse, resulting from either direct dumping or transport from Clearview Landfill. Sediment in Darby Creek downstream of the landfill was classified as medium/very fine sand and silt. In Tinicum Marsh, the sediment was classified as silt, typical of a low-energy marsh environment.

Average sediment PAH concentrations in various grain sizes are listed in Table 4-9 and plotted on Figure 4-26. In general, the larger the grain size, the higher the concentration of PAHs. This could be related to the larger grains likely being exposed to contamination longer than smaller grains. Low concentration of PAHs in coarse sand fraction of sediment is simply because this fraction is not common in Darby and Cobbs Creeks. Therefore, sediment grain size plays an important role in PAHs distribution in sediment.

#### 4.2.3.2 Tidal Study Results

The water volume exchange during an average tidal cycle at the S. 84<sup>th</sup> Street bridge over Darby Creek was conceptually analyzed. This analysis generally indicated that approximately six times more water flows downstream (ebb tide) than upstream (flood tide) during a typical tidal cycle, and that flow velocities were correspondingly smaller during the flood-tide part of the cycle. Based on the average flow of water, approximately 7.2 million cubic feet of water is discharged to Tinicum Marsh from Darby Creek, while approximately 1.2 million cubic feet of water flows toward S. 84<sup>th</sup> Street during the tidal cycle. The background information on this analysis is available in Appendix N.

Tidal influences were measured in the field during a tidal study performed in January 2003 at six staff gages (see Figure 3-9 for locations). Figure 4-27 shows the water levels measured at six staff gages during a study period. These tidal measurements indicated a tidal amplitude of at least 4.5 feet at the staff gage (SG1) in Tinicum Marsh near I-95, with a strong decrease in tidal amplitude to a minimum of approximately 0.3 feet in Cobbs Creek at 78<sup>th</sup> Street (upon a complete tidal cycle). During the study, qualitative, visual observations of tidal amplitude indicated that Cobbs Creek near the northern end of Clearview Landfill had smaller tidal amplitude. However, a 1- to 2-foot high tidally-influenced wave occurred and moved rapidly upstream in Darby and Cobbs Creeks as the flood tide began.

Given the relatively smaller flood-tide water volume under normal tide conditions, the majority of sediment transport is expected under ebb-tide conditions when normal downstream flow occurs in Darby and Cobbs Creeks. In addition, larger particle sizes are expected to be carried by faster-moving water during the ebb tide, while only minor sediment transport is expected to occur under flood-tide conditions.

### 4.3 Soil Evaluation

Soil sampling results from 127 Geoprobe borings are presented in Tables 4-10 through 4-13 for SVOCs, pesticides/PCBs, metals, and dioxin analyses, respectively. Tables 4-14 and 4-15 include background soil sampling results. The sampling results were compared to USEPA Region 3 RBCs (October, 2007) for



residential soil and BTAG Screening Benchmarks for soil (USEPA, 1995) for discussion purposes. See Appendix O for the complete soil sample analytical results.

#### 4.3.1 Surface Soils

Metals, SVOCs, pesticides, and PCBs were the principal classes of contaminants detected in surface soils collected from the Geoprobe borings. Figures 4-28 through 33 display the concentrations of selected key substances in surface soil – arsenic, lead, benzo(a)pyrene, total PCBs, total pesticides, and dioxins, respectively.

Figure 4-28 shows that arsenic was encountered above its RBC for residential soil (0.43 mg/kg) in all surface soil samples (except one location GP037), but below its ESV (328 mg/kg). The highest arsenic concentration was 42.2 mg/kg at location GP251 in the southern industrial area where several businesses are currently being operated. This area generally had high levels of arsenic in surface soil, as seen at GP215 (14.1 mg/kg), GP231 (13 mg/kg), GP237 (12.1 mg/kg), and GP239 (18.5 mg/kg). The second highest detection was 20.3 mg/kg at GP102 in the northern part of the Eastwick neighborhood outside the landfill. Elevated arsenic was also detected at GP032 (15.2 mg/kg) in the northern part of City Park.

Figure 4-29 shows distribution of lead in surface soil samples. High concentrations were clustered in the northern part of City Park and the southern industrial area in Clearview Landfill.

Figure 4-30 shows benzo(a)pyrene concentrations detected in the surface soil samples. In general, elevated PAHs were detected in surface soil within the landfill footprint, but were also scattered in the Eastwick neighborhood outside the landfill footprint. Elevated PAH concentrations were also clustered in the northern part of the City Park and in the southern industrial area where oily wastes were encountered during drilling.

Figure 4-31 displays the distribution of total PCBs in surface soil samples. PCBs were scattered among the surface soil samples, mostly inside the landfill footprint. Significant PCB concentrations were found in the southern industrial area where leaky transformers were found (i.e., near GP239). High PCB concentrations were also found in the enclosed low area adjacent to the industrial area (i.e., GP034, GP035, and GP037). In addition, elevated PCB concentrations were clustered in the northern open field in the City Park area.

Current operations in the southern industrial area are producing visible spills/leaks as observed by field personnel during the RI. Activities in this area are coincident with substantial detected contamination. High PAH contamination was observed in the following areas:

- Transformer recycling area where one visibly leaking transformer carcass was reported by field personnel to USEPA.
- Truck repair area, including the enclosed low drainage area that drains most of the industrial area. Note that two shallow soil samples (GP215 and 239) in this area contained PCB levels above RCRA characteristic hazardous waste concentrations (i.e., > 50,000 µg/kg of PCBs).
- Area with waste oil staining on the ground surface where free-product oil was found in boreholes.
- Drum recycling area where spilled/leaked substances were observed on the ground.

Total pesticides (sum of all pesticides detected) are shown on Figure 4-32. Pesticides were wide-spread throughout OU-1, but they were detected more frequently and at higher concentrations in Clearview Landfill than City Park or Eastwick neighborhood area.

Surface soils in five locations within the southern industrial area of the landfill were collected and analyzed for dioxins. The results are shown on Figure 4-33.

Concentrations of the following analytes exceeded their respective RBCs for residential soil or ESVs in at least one surface soil sample:

#### Metals

- |             |            |
|-------------|------------|
| • Aluminum  | • Antimony |
| • Arsenic   | • Barium   |
| • Beryllium | • Cadmium  |
| • Chromium  | • Copper   |
| • Iron      | • Lead     |
| • Manganese | • Mercury  |
| • Nickel    | • Selenium |
| • Silver    | • Thallium |
| • Vanadium  | • Zinc     |

#### SVOCs

- |                            |                           |
|----------------------------|---------------------------|
| • Acenaphthylene           | • Anthracene              |
| • Benzo(a)pyrene           | • Benzo(a)anthracene      |
| • Benzo(b)fluoranthene     | • Benzo(g,h,i)perylene    |
| • Benzo(k)fluoranthene     | • Chrysene                |
| • Dibenzo(a,h)anthracene   | • Fluoranthene            |
| • Fluorene                 | • Indeno(1,2,3-C,D)pyrene |
| • 4-methylphenol(p-cresol) | • Naphthalene             |
| • Phenanthrene             | • Pyrene                  |

#### Pesticides/PCBs

- |                           |                                                                     |
|---------------------------|---------------------------------------------------------------------|
| • Aldrin                  | • Alpha-chlordane                                                   |
| • Beta-BHC                | • Dieldrin                                                          |
| • Endrin                  | • Gamma-chlordane                                                   |
| • Heptachlor Epoxide      | • Methoxychlor                                                      |
| • DDD                     | • DDE                                                               |
| • DDT                     | • PCB-1242 (Aroclor 1242)                                           |
| • PCB-1254 (Aroclor 1254) | • PCB-1260 (Aroclor 1260) – the most frequently detected PCB isomer |
| • PCB-1268 (Aroclor 1268) |                                                                     |

The following discussion provides details regarding the occurrences of contamination in surface soil (six inches deep) within each investigative area:

#### *Clearview Landfill*

**Metals** – All eighteen metals listed above were detected at concentrations above their respective RBCs or ESVs in the surface soil samples collected from Clearview Landfill. Arsenic was detected above its RBC at all sampling locations (Figure 4-28), but below its ESV (328 mg/kg). Five sampling locations (GP215, GP231, GP237, GP239, and GP251) contained elevated arsenic concentrations up to 42.2 mg/kg, all of which are greater than PADEP residential soil screening value of 12 mg/kg, but less than non-residential

soil screening value of 53 mg/kg. All five locations were in the southern portion of the landfill within the currently active industrial businesses (see Figure 2-3).

As shown on Figure 4-29, lead was detected at many sampling locations within the Clearview Landfill. Twelve locations exhibited lead in surface soil at concentrations greater than 400 mg/kg, which is the soil lead level for residential use set by USEPA Office of Solid Waste Directive. Of these, four locations had lead concentrations greater than 1,000 mg/kg (1,840 mg/kg at GP215; 1,410 mg/kg at GP231; 1,130 mg/kg at GP239; and 1,060 mg/kg at GP248), all in the southern industrial area.

Other metals detected at elevated concentrations include aluminum ranging from 1,810 (GP244) to 24,900 mg/kg (GP226), and iron ranging from 5,510 (GP244) to 152,000 mg/kg (GP251).

VOCs – No VOCs were detected at concentrations exceeding the RBCs or ESVs in surface soils within the landfill footprint.

SVOCs – All SVOCs (all PAHs) listed above, except 4-methylphenol, were detected above their respective RBCs or ESVs (100 µg/kg for all PAHs) throughout Clearview Landfill in surface soil. As shown on Figure 4-30, concentrations of benzo(a)pyrene (the primary risk driver among the detected PAH compounds) ranged from 260 to 11,000 µg/kg, with two high concentrations in the center (11,000 µg/L at GP219) and western bank (8,000 µg/L at GP227) of the landfill. Medium to high PAH concentrations (90 – 4,000 µg/L) were clustered within the southern industrial area, particularly near visible ground staining in the truck repair and auto parts businesses.

PCBs – PCBs were detected in surface soil within Clearview Landfill. The most commonly detected PCB was PCB-1260 (Aroclor 1260). As with other contaminants, very high concentrations of total PCBs (Figure 4-31) were detected within the southern industrial area, specifically at GP215 (51,000 µg/kg) and at GP239 (280,000 µg/kg). It should be noted that shortly after sampling had been performed at this area, a transformer carcass was found to be overturned and leaking oil immediately uphill of sampling location GP239. Two additional PCB isomers, PCB-1242 (Aroclor 1242) and PCB-1268 (Aroclor 1268), were detected at levels exceeding their respective RBCs (320 µg/kg) or ESVs (100 µg/kg) within the landfill, but not at the frequency of PCB-1260. Surface soil in the low area of the southern part of City Park (near GP034 and 035) was also impacted by PCBs, most likely transported from the landfill (see the City Park section below.)

Pesticides – Concentrations of total pesticides in Clearview Landfill ranged from 2.6 to 24,702 µg/kg (Figure 4-32). The highest concentration was detected at GP239, located in the currently active southern industrial area. Surface soil in this location was detected with many pesticides at concentrations that exceeded their RBCs or ESVs, including alpha chlordane, beta BHC, DDD, DDE, DDT, dieldrin, endrin, gamma chlordane, heptachlor epoxide, and methoxychlor. These pesticides were detected at low concentrations (below RBCs or ESVs) in surface soil throughout the landfill.

Dioxins – For dioxin analysis, surface soil samples were collected from three locations (GP249, 250, and 251) within the southern industrial area of the landfill in January 2006 (Figure 4-33). Dioxin was analyzed in soil samples in these locations because of the incinerator operation in this area that was reported by former site workers, and concerns about recreational use of City Park. Total dioxin concentrations (total PCDDs and PCDFs) in surface soils were 1.2041 pg/g at GP249; 0.7521 pg/g at GP250; and 0.3175 pg/g at GP251. None of these total dioxin concentrations exceeded the USEPA Region 3 RBC of 4.3 pg/g, ESV of 10,000 pg/g, or the Removal Action levels of 430 pg/g.

### **Special Note on Dioxin Nomenclature Used in this Report**

Dioxins and furans are presented in this report using a “Total EQuivalents” (TEQ) system, in which the total (sum) concentration of polychlorinated dibenzodioxins (dioxins, or PCDDs) and polychlorinated dibenzofurans (furans, or PCDFs) are reported as “Total Equivalents” to the specific dioxin compound 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD). Comparisons of actual measured dioxin concentrations throughout this report are made using this “Total Equivalents” system. The EPA Region 3 RBCs reported for 2,3,7,8-TCDD for groundwater, soil, and sediment will be used for comparison with site data. For simplicity, “dioxin” will be used hereafter to refer to the total sum of dioxins and furans under this naming convention.

#### *City Park*

**Metals** – Arsenic was detected above its RBC (0.43 mg/kg) or ESV (328 mg/kg) throughout the surface soil samples collected from the City Park area. Relatively high arsenic concentrations (between 7 to 15.2 mg/kg) were detected at four locations (GP026, 029, 032, 035, and 081). The highest concentration (15.2 mg/kg, which is greater than PADEP residential soil screening value of 12 mg/kg) was detected at GP032 located in the northern part of the City Park. Lead was widely detected throughout the City Park area. Five locations exhibited lead concentrations over 400 mg/kg (the soil lead level for residential use set by USEPA Office of Solid Waste Directive). Of these, the highest concentration of lead (8,540 mg/kg) in surface soil was detected at GP021 in the northern open field of the City Park.

**VOCs** – No VOCs were detected at levels exceeding their respective RBCs or ESVs within the City Park area.

**SVOCs** – Fifteen SVOCs in surface soils were detected in exceedance of the RBCs or ESVs throughout the City Park. The primary SVOCs were PAHs, including acenaphthylene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, dibenz(a,h)anthracene, indeno(1,2,3-C,D)pyrene, and phanthrene. Of these, concentrations of benzo(a)pyrene were plotted on Figure 4-30 and ranged from non-detect to 4,000 µg/kg. Significant concentrations of benzo(a)pyrene were detected throughout City Park. The highest concentration was detected at GP018 (4,000 µg/kg) located in the northern part of City Park. High concentrations of benzo(a)pyrene were also detected in sampling locations GP031 (3,900 µg/kg) and 034 (2,400 µg/kg), which were located near the Angelo Place residences and downgradient of the southern industrial area in the landfill, respectively (boring GP034 also encountered probable free-product oil).

**PCBs** – PCBs were detected in surface soil throughout the City Park area. The most commonly detected was PCB-1260 (Aroclor 1260). As shown on Figure 4-31, concentrations of total PCBs in surface soil ranged from non-detect to 37,000 µg/kg. The surface soil in the low area (see Figure 3-12) of southern City Park might receive drainage from the southern industrial area of the landfill with elevated total PCB concentrations, resulting in high PCB contamination (e.g., 37,000 µg/kg at GP034; 16,600 µg/kg at GP035; and 6,100 µg/kg at GP037). In addition, elevated PCB concentrations were detected at GP021 (1,700 and 390 mg/kg for PCB-1242 and 1260, respectively) located within the northern area of City Park, along with high levels of other contaminants. In the 1958 aerial photo (Figure 1-3), an industrial facility appeared northwest of Clearview Landfill along Cobbs Creek. A probable fill area containing dark-toned material (DTM) was located east of the facility, presently an open field in City Park. In the 1965 aerial photo (Figure 1-3), junked autos and debris were visible, and a large pile of DTM, a deep pit containing dark standing liquid, and a crane were clearly evident in this northern part of City Park. The

recreational area (e.g., tennis court, basket ball field, etc.) appeared next to this area in 1983 aerial (Figure 1-3).

**Pesticides** – Pesticides, including aldrin, dieldrin, endrin, gamma chlordane, methoxychlor, DDD, DDE, and DDT, were detected at levels exceeding their RBCs or ESVs (100 µg/kg) within the City Park area. As shown on Figure 4-32, pesticides was found mostly in the southern part of City Park (total pesticide concentration of 2,305 µg/kg at GP034, 216 µg/kg at GP035, and 328µg/kg at GP037). In addition, two other locations GP021 and GP090 exhibited total pesticide concentrations of 199 and 297 µg/kg, respectively.

#### *Eastwick Neighborhood*

**Metals** – Arsenic was detected above its RBC (0.43 mg/kg) or ESV (328 mg/kg) in surface soil throughout the Eastwick neighborhood area, at concentrations ranging from 2.0 to 20.3 mg/kg. The highest concentration was detected at GP102, located in the northern portion of the Eastwick neighborhood (an unusual outlier value not associated with nearby contaminants), exceeding PADEP residential soil screening values for arsenic of 12 mg/kg. Two sampling locations, GP002 (in the northern part) and GP006 (a few blocks east of the Angelo Place residences), also exhibited concentrations of arsenic at 8.7 and 10 mg/kg, respectively. Lead was detected in surface soil throughout the Eastwick neighborhood at concentrations greater than its ESV (0.01 mg/kg), ranging from 4.6 (at GP093) to 124 mg/kg (at GP012). No location exhibited lead concentrations that exceeded the lead level for residential use (400 mg/kg) per USEPA Office of Solid Waste Directive.

**VOCs** – No VOCs were detected at concentrations exceeding the RBCs or ESVs in surface soil within the Eastwick neighborhood area.

**SVOCs** – Many SVOCs were detected in surface soil throughout the Eastwick neighborhood area. The primary SVOCs detected in exceedance of the RBCs or ESVs were PAHs, including acenaphthrene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, dibenz(a,h)anthracene, indeno(1,2,3-C,D)pyrene, and phenanthrene. Concentrations of benzo(a)pyrene (the primary risk-driver among the detected PAH compounds) ranged from non-detect to 3,600 µg/kg. Substantial concentrations of benzo(a)pyrene were detected at four locations—2,500 µg/kg at GP007; 3,600 µg/kg at GP106; 2,800 µg/kg at GP108; and 2,200 µg/kg at GP109. Of these, GP106, 108, and 109 were located near the eastern boundary of the City Park area, whereas GP007 was located in the lower southern part of the neighborhood.

**PCBs** – PCBs in surface soil were analyzed throughout the Eastwick neighborhood area. Only four locations (GP008, 017, 105, and 109) were detected with PCBs in surface soil. Of these, surface soil at GP109 located near the City Park area exhibited PCB-1260 at a concentration of 370 µg/kg, exceeding its RBC value (320 µg/kg) and ESV (100 µg/kg).

**Pesticides** – Pesticides were detected at low concentrations in the Eastwick neighborhood, as is typical in an urban setting. However, no locations within the Eastwick neighborhood exhibited concentrations that exceeded their RBCs or ESVs (100 µg/kg).

#### 4.3.2 Subsurface Soils

Subsurface soil samples were collected from the same locations as the surface soil samples. Two samples were collected from each location: one from medium depths ranging from 0.5 to 9 feet, and the other from deep depths ranging from 6 to 32 feet, depending on location.

Similar to the findings in the surface soil analysis, metals, SVOCs, and pesticide/PCBs were the principal classes of contaminants detected in the subsurface soils. Tables 4-10 through 4-12 summarize the analytical results of SVOCs (PAHs), pesticide/PCBs, and metals in subsurface soil, respectively. Figures 4-34 through 4-38 show the distribution and concentrations of arsenic, lead, benzo(a)pyrene, total PCBs, and total pesticides, respectively. The distribution of dioxins in subsurface soil is shown on Figure 4-33.

The concentrations of the contaminants in the subsurface samples were generally lower than their concentrations in the surface soils and decreased with depth, except for several SVOCs, mostly PAHs (higher in subsurface soil).

In the subsurface soil samples, the analytes with concentrations that exceeded their respective RBCs or ESVs in at least one sample were as follows:

##### SVOCs

- Acenaphthylene
- Benzo(a)pyrene
- Benzo(b)fluoranthene
- Benzo(k)fluoranthene
- Bis(2-chloroethyl)ether
- Dibenz(a,h)anthracene
- 2,4-Dinitrophenol
- Fluorene
- 4-Methylphenol(p-cresol)
- 4-Nitrophenol
- Phenanthrene
- Anthracene
- Benzo(a)anthracene
- Benzo(g,h,i)perylene
- Bis(2-ethylhexyl)phthalate
- Chrysene
- 2,4-Dimethylphenol
- Fluoranthene
- Indeno(1,2,3-C,D)pyrene
- Naphthalene
- Pyrene
- 1,2,4,5-Tetrachlorobenzene

##### Pesticides/PCBs

- PCB-1242 (Aroclor 1242)
- PCB-1254 (Aroclor 1254)
- DDD
- DDT
- PCB-1260 (Aroclor 1260)
- Dieldrin
- DDE

##### Metals

- Aluminum
- Antimony
- Beryllium
- Copper
- Lead
- Mercury
- Silver
- Zinc
- Arsenic
- Chromium
- Iron
- Manganese
- Nickel
- Vanadium

The following discussion provides details regarding the occurrences of contamination in subsurface soil within each investigative area of Clearview Landfill, City Park, and Eastwick neighborhood. The USEPA Region 3 RBCs for residential soil and BTAG Screening Benchmarks (ESVs) were used for discussion purposes.

#### *Clearview Landfill*

**Metals** – Subsurface soil for metal analysis was collected only in City Park in 2003. Therefore, no subsurface soil samples in Clearview Landfill were analyzed for metals.

**VOCs** – No VOCs were detected in concentrations exceeding the RBCs or ESVs in subsurface soils within the landfill.

**SVOCs** – Many SVOCs (primarily PAHs) were detected above the RBCs or ESVs in subsurface soils throughout Clearview Landfill. Of these, concentrations of benzo(a)pyrene are shown on Figure 4-36 to display the general distribution of these SVOCs.

The highest concentration of benzo(a)pyrene (110,000 µg/kg) was detected in the 10-foot deep soil sample collected from GP218 near Darby Creek. Other PAHs were also detected in this sample at elevated concentrations above their respective RBCs or ESVs, including benzo(a)anthracene (130,000 µg/kg); benzo(b)fluoranthene (29,000 µg/kg); benzo(g,h,i)perylene (57,000 µg/kg); benzo(k)fluoranthene (51,000 µg/kg); indeno(1,2,3-C,D)pyrene (57,000 µg/kg); and phenanthrene (310,000 µg/kg). In addition, a 5-foot deep soil sample from this boring location revealed elevated levels of PAHs, including 2,300 µg/kg of benzo(a)pyrene. The locations with significant concentrations of PAHs in subsurface soil were typically located near Darby and Cobbs Creeks where leachate seeps were observed, as well as in the southern industrial part of the landfill.

**Pesticides** – Pesticides were not analyzed for subsurface soil in the landfill.

**PCBs** – Several PCBs were detected in subsurface soil throughout the Clearview Landfill area, but only one isomer of PCB (PCB-1260) was detected in exceedance of the RBC (320 µg/kg) or ESV (100 µg/kg). A significantly elevated concentration of PCB-1260 (2,100 mg/kg) was detected in the 0.5-foot deep soil sample at GP245 near the access road on the landfill. As with the majority of other contaminants, high PCB concentrations in subsurface soil within the landfill were mostly detected in the southern industrial area. PCBs in subsurface soils exceeded the RCRA Characteristic Hazardous Waste concentration of 50,000 µg/kg at three locations—GP215 (91,000 µg/kg), GP234 (120,000 µg/kg), and GP245 (2,100 mg/kg), all in the southern part of the landfill.

**Dioxins** – As shown on Figure 4-33, dioxins were detected in subsurface soils collected from two locations within the southern industrial area in January 2006. Total dioxin concentrations (total PCDDs and PCDFs) in subsurface soils (10 feet deep) were 8.87 pg/g at GP241 and 47.26 pg/g at GP245, which exceeded the USEPA Region 3 RBC for residential soil of 4.3 pg/g, but less than ESV (10,000 pg/g).

#### *City Park*

**Metals** – Metal analysis for subsurface soil was limited to soil boring locations GP042 through GP049 and GP051 through GP058. Arsenic was detected above its RBC in all subsurface soil samples collected from these locations, and their concentrations ranged from 0.93 to 7.7 µg/kg, which are less than ESV (328 µg/kg). The highest arsenic concentration was detected in the 5-foot deep soil sample from GP047

near the southeastern edge of the historical landfill footprint. Other subsurface sampling locations with arsenic levels above its RBC included soil borings along walking paths in City Park (GP049, 051, 052, 053, 054, 055, and 056), and several soil borings near tennis and basketball courts (GP044, 045, 057, and 058) in the northern part of City Park.

Lead was detected in subsurface soil at 16 boring locations throughout the City Park area (Figure 4-35). The highest concentration of lead (279 mg/kg) was detected at GP047 near the townhomes in the southern part of City Park. Other metals detected above their respective RBCs or ESVs in subsurface soil sampling locations include antimony, beryllium, chromium, nickel, vanadium, and zinc.

VOCs – VOCs were not detected at levels exceeding their RBCs or ESVs in subsurface soils within the City Park area.

SVOCs – Many SVOCs were detected in subsurface soils throughout the City Park area, exceeding their respective RBCs or ESVs. Concentrations of benzo(a)pyrene in subsurface soil are shown on Figure 4-36 to present the general distribution of SVOCs throughout City Park. Concentrations of benzo(a)pyrene ranged from non-detect to 13,000 µg/kg; its highest concentration was detected at GP094, which was located on the residential properties near the Angelo Place. Two other locations (GP034 and GP247) near the southern industrial area of the landfill also contained significant concentrations of benzo(a)pyrene in the subsurface soil samples (2,000 µg/kg at GP034 and 1,200 µg/kg at GP247).

Pesticides – Pesticides were detected in subsurface soil throughout City Park. The highest concentration of total pesticides (primarily DDD and DDE) was detected in the deep soil sample collected from location GP018 (1,670 µg/kg) in the northern part of City Park near the open field. The 16-foot deep soil sample from soil boring GP093 within the residential properties near Angelo Place exhibited elevated concentrations (above 100 µg/kg of ESVs) of pesticides such as DDD (300 µg/kg), DDE (120 µg/kg), and DDT (220 µg/kg), but less than their respective RBCs.

PCBs – In City Park, only two subsurface samples were detected with PCB-1260 at concentrations exceeding its RBC (320 µg/kg) or ESV (100 µg/kg) – the medium deep (10 feet – 120 µg/kg) and deep (12 feet – 400 µg/kg) samples at GP018 near Cobbs Creek in the northern portion of the City Park. This location also contained high concentrations of pesticides as discussed above. The medium deep soil samples (0.5 to 9 feet deep) collected in the southern part of City Park also exhibited the presence of PCBs (up to 1,000 µg/kg). This area receives discharge from the southern industrial area of the Clearview Landfill. However, none of the PCB levels exceeded the RBC.

Note that very high vapor concentrations, free-product oil, and visually-evident contamination were detected in subsurface soil in the northern part of the City Park. However, the analytical results for subsurface soil indicated very low concentrations of contaminants and did not reflect these conditions.

### *Eastwick Neighborhood*

Metals – No metals were analyzed for subsurface soil in the Eastwick neighborhood area.

VOCs – No VOCs were detected in concentrations exceeding their respective RBC or ESVs within subsurface soil in the Eastwick neighborhood area.

SVOCs – Many SVOCs were detected in subsurface soil in the Eastwick neighborhood, but at relatively low concentrations. As shown on Figure 4-36 with benzo(a)pyrene as representative of SVOCs, the



highest concentration (2,000 µg/kg) was detected in a 10-foot deep sample from GP002 in the northern part of the neighborhood; however, benzo(a)pyrene was not detected in the soil sample at a depth of 5 feet directly above in the same boring. Locations of detection and concentrations of benzo(a)pyrene (and other SVOCs) were very sporadic in subsurface soil in the Eastwick neighborhood.

PCBs – No PCBs were detected in subsurface soil within the Eastwick neighborhood.

Pesticides – Pesticides were not detected in most subsurface soil samples collected from this area. However, one abnormally high concentration (5,600 µg/kg of DDD) was detected in the deep subsurface soil sample at GP002, located near the northeastern boundary of the City Park. High concentrations of PAHs, such as benzo(a)pyrene, were also detected in the deep soil sample from this location.

#### 4.3.3 Extent of Non-Aqueous Phase Liquids (NAPLs)

Free-product oil or non-aqueous phase liquids (NAPLs) were noted during drilling of several borings and wells near Clearview Landfill. NAPLs are immiscible hydrocarbons in the subsurface, which can partially dissolve into water at very slow rates and form a visible, separate oily phase in the subsurface. They can be either lighter than water (LNAPL) or denser than water (DNAPL). The locations of borings and monitoring wells containing NAPLs are indicated on Figure 4-39. See Appendix B for further information on subsurface materials encountered at these locations. These borings can be grouped as follows:

- **Buried Stream Channel in the City Park.** Soil borings GP027, GP031, GP035, and GP050 all had signs of floating LNAPL and DNAPL (free-product) within either water samples or soil samples removed from the boreholes during drilling. These borings were located approximately in the historic stream channel, which existed prior to the commencement of landfilling activities.
- **Northern End of the City Park Area.** Oily soil and water were observed at borings GP032 and GP033 located in the northern part of the City Park area, approximately where pits and junked autos were observed on historical aerial photographs (see Figure 1-3). Surface and subsurface soils in this area were found to have high contamination, as discussed previously.
- **Clearview Landfill.** Free-product oil was found at monitoring well MW10 located in the central portion of Clearview Landfill and at soil borings GP218, 224, and 227 near the western edge of the landfill. Because of the thick depth of waste disposal in this area, the actual source of the free-product noted in the borings was difficult to determine. However, this area is of particular interest because of the amount of recharge and groundwater mounding noted in the groundwater mapping and modeling. Groundwater flow likely carries contamination to multiple directions (from this area toward Darby and Cobbs Creeks and toward City Park/Eastwick neighborhood).
- **Southern Industrial Area in Clearview Landfill.** Seven soil borings (GP231, 233, 237, 238, 239, 244, and 251) and two wells (MW04 and 12) contained evidence of free-product oil, and were located within this active industrial area. As discussed previously, the majority of the highest concentrations of contaminants were detected in this area. Free-product oil in this area could be the result of previous dumping, current operations, or both. Based on the direction of groundwater flow, it is likely that free-product oil detected in this area flows toward Darby Creek. Furthermore, an oily sheen was observed in seeps from the banks of Darby Creek adjacent to this area.

#### 4.3.4 Background Soil Quality

In April 2007, background soil samples were collected from off-site locations, including the northern part of the John Heinz NWR located on the east bank of Darby Creek across the Industrial Drive properties and the Sun Oil Darby Creek Tank Farm, as well as the Korman Suites property located on Lindbergh Boulevard south of S. 84<sup>th</sup> Street (see Figure 3-10 for locations). Soil samples were collected from these areas to determine the concentrations of substances in soil, which can be attributed to natural and anthropogenic non-point sources within a local background area, without any contribution from Clearview Landfill.

As Figure 3-10 shows, soil samples were collected from 11 locations using a hand auger, including 7 within the Korman Suites property (PSS-13, 15, 18, 19, 27, 29, and 30) and 4 in the northern part of the John Heinz NWR (PSS-07 through 10). Two samples were collected from each location—a shallow subsurface sample (0 to 6 inches deep) and a deep subsurface sample (12 to 24 inches deep). The analytical results of substances detected in soil within the Korman Suites property and John Heinz NWR are summarized in Tables 4-15 and 4-16, respectively.

A statistical analysis, comparing background soil data and site-related soil data, was performed as part of the HHRA. Refer to Appendix U, Part 4 for the detailed information.

##### Korman Suites

According to the United States Department of Agriculture (USDA), Natural Resources Conservation Service (NRCS)'s soil survey for Delaware and Philadelphia Counties, soil in this area is classified as urban lands, consisting of mostly silt loam. The soil in sampling locations within the Korman Suites property consisted primarily of common urban fill material (e.g., bricks and debris), as anticipated (see Section 2.4). Soil was most likely brought into this area to level the ground when property development commenced.

Soil samples from all 7 sampling locations were detected with substances that were found in Clearview Landfill, but at much less significant concentrations. However, these substances were detected in the soil samples at concentrations exceeding either their respective RBCs or ESVs.

All but six of the 24 inorganics analyzed were detected in soil samples with concentrations exceeding their respective RBCs or ESVs. The six inorganics, including barium, cadmium, calcium, sodium, potassium, and magnesium, were either detected less than their screening values or have no screening values. Concentrations of arsenic in soil samples varied between 2.4 and 11.4 mg/kg (highest in a deep soil sample from PSS-15). Both surface and subsurface samples at all 7 sampling locations were detected with nine metals at elevated concentrations (exceeding their respective RBCs or ESVs), including aluminum, arsenic, chromium, copper, iron, lead, nickel, vanadium, and zinc.

PAHs, such as benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, fluoranthene, and pyrene were detected in the soil samples, exceeding their respective RBCs or ESVs. Of these, benzo(a)pyrene was detected in both shallow and deep soil samples from all 7 locations at concentrations greater than its RBC (22 µg/kg), but less than ESV (100 µg/kg). Other PAHs were also detected in the surface soil or subsurface soil samples from five locations (PSS-13, PSS-19, PSS-27, PSS-29, and PSS-30) at concentrations above their respective RBCs or ESVs.

Aroclor-1260 was detected in both shallow and deep soil samples from two sampling locations (PSS-15 and 30). However, their concentrations were below the RBC (320 µg/kg) and ESV (100 µg/kg).

### John Heinz NWR

Based on soil sample logs, the northern part of the John Heinz NWR appears to have received fill material since bricks and other debris were found in soil sampling locations PSS-7, 8, and 9. However, the northernmost sampling location, PSS-10 contained native sand material.

Similar to the findings in the Korman Suites property, concentrations of many metals exceeding their RBCs or ESVs were detected in both shallow and deep soil samples collected from all four sampling locations. Arsenic was detected at concentrations ranging from 1.2 mg/kg (in a deep soil sample from PSS-010) to 42.5 mg/kg (in a surface soil sample from PSS-08). In addition to arsenic, antimony (53.3 mg/kg) in a deep soil sample collected from PSS-08 exceeded its RBC (31 mg/kg) and ESV (0.48 mg/kg); and iron concentrations in both shallow (33,600 mg/kg) and deep soil samples (122,000 mg/kg) from this location exceeded its RBC (55,000 mg/kg) or ESV (12 mg/kg). Three pesticides, dieldrin, DDD, and DDE, were detected at their RBC levels (40 µg/kg for dieldrin) or ESV levels (100 µg/kg for all three) in the deep soil sample from PSS-08.

Twelve PAHs (anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, fluoranthene, fluorene, indeno(1,2,3-C,D)pyrene, phenanthrene, and pyrene ) with exceedance of their respective RBCs or ESVs were detected in the soil samples collected in the northern part of the John Heinz NWR. Benzo(a)pyrene concentrations exceeding its RBC were detected in both shallow and deep soil samples from all four sampling locations, and ranged from 110 µg/kg (in a surface soil from PSS-010) to 630 µg/kg (in a deep soil sample from PSS-08). Benzo(b)fluoranthene (1,100 µg/kg) exceeded its RBC (220 µg/kg) or ESV (100 µg/kg) in both surface and deep soil samples collected from four locations, except in a surface soil sample from PSS-09. Dibenzo(a,h)anthracene was detected in both shallow and deep soil samples from two locations, PSS-07 and 08, but did not exceed its ESV (100 µg/kg). Indeno(1,2,3-cd)pyrene concentrations exceeded its ESV (100 µg/kg) in the deep samples from PSS-9 (160 µg/kg) and PSS-10 (100 µg/kg), and in both shallow and deep soil samples from PSS-7 and 8 (230, 110, 250, and 370 µg/kg, respectively).

Aroclor 1260 was detected in both shallow and deep soil samples at two locations (PSS-08 and 10). Of these samples, only the Aroclor 1260 concentration (850 µg/kg) in a deep soil sample from PSS-08 exceeded its RBC (320 µg/kg) and ESV (100 µg/kg).

In summary, many PAHs and metals were detected at concentrations greater than their respective RBCs or ESVs in the background soil samples. The highest concentrations of contaminants were detected at sampling location PSS-8 south of the USFWS maintenance building, while the lowest concentrations were detected at sampling location PSS-10 located between the maintenance building and the S. 84<sup>th</sup> Street bridge.

#### 4.4 Groundwater Evaluation

Previous groundwater sampling indicated that groundwater in the Clearview Landfill area was contaminated, and the potential existed for migration of groundwater contamination toward the Eastwick neighborhood or into Darby and Cobbs Creeks. However, these sampling events were limited in extent. As a result, a significant groundwater sampling program was implemented for the RI.

Based on the analytical results of the groundwater samples from monitoring wells and borings, it was found that groundwater quality in the study area has been impacted by organic and inorganic

contaminants originating from wastes in Clearview Landfill. Contaminated groundwater exists below Clearview Landfill, in the City Park area on the east side of the landfill, and in the Eastwick neighborhood. Groundwater samples were collected during sampling events in 2003, 2004, and 2006. The groundwater samples from shallow monitoring wells MW-01S, MW-02, MW-03, MW-04, MW-05S, MW-06, MW-07S, MW-08, MW-09, MW-10, MW-11, MW-12, and the Geoprobe borings represent shallow groundwater quality at/above the water table. Groundwater samples from the deep monitoring wells MW-01D, MW-05D and MW-07D represent groundwater quality below the water table.

Groundwater analyses were conducted to determine the nature, extent and migration potential of site-related pollutants. Tables 4-16 through 4-19 summarize concentrations of these pollutants that were above the RBCs for tap water, including VOCs, SVOCs, pesticide/PCBs, and TAL metals, respectively. Currently, groundwater at the site is not used as potable water; however, the RBCs for tap water were used to conservatively discuss groundwater contamination in accordance with USEPA's policy for future use of groundwater as potable water. The analytical results for all groundwater samples are compiled in Appendix P. Well sampling logs with field measurement parameters are provided in Appendix D. Groundwater samples from the Geoprobe borings GP03, 04, 05, 06, 23, 39, 40, 44, 45, 56, and 80 were not collected during the 2003 sampling event because these borings were dry. For that same reason, the groundwater samples from GP88, 94, 95, 101, 103, and 104 were also not collected during the 2004 sampling event.

Several analytes were selected to represent the various analyte groups such as metals, SVOCs (PAHs), and VOCs. Figures 4-40 through 4-45 show concentrations of the selected analytes such as arsenic, benzo(a)pyrene, benzene, chlorobenzene, total PCBs, and total pesticides, respectively, which represent the distribution of pollutant groups that were commonly found in groundwater.

Figure 4-40 shows the distribution of arsenic in the groundwater samples. Total arsenic concentrations ranged from non-detect to 241  $\mu\text{g/L}$ , and appeared to be pervasive in the shallow groundwater and extended into the Eastwick neighborhood outside the historical footprint of the landfill.

Benzo(a)pyrene is generally found in coal tar, automobile exhaust fumes (especially from diesel engines), all smoke resulting from the combustion of organic material, and charbroiled food. It is a product of incomplete combustion. Benzo(a)pyrene concentrations in groundwater samples ranged from non-detect to 8  $\mu\text{g/L}$ . As shown on Figure 4-41, relatively high concentrations of benzo(a)pyrene in groundwater (RBC = 0.003  $\mu\text{g/L}$ ) were generally limited to the southern industrial area in Clearview Landfill. However, groundwater in the northern part of City Park and the western bank of the landfill was also detected with elevated benzo(a)pyrene levels.

Figures 4-42 and 4-43 show the distribution of benzene and chlorobenzene, respectively. Benzene has been widely used as a fuel additive to increase the octane rating, while chlorobenzene has been used as an industrial solvent. Concentrations of benzene ranged from non-detect to 18  $\mu\text{g/L}$ , and concentrations of chlorobenzene ranged from non-detect to 190  $\mu\text{g/L}$  at GP243. High concentrations of benzene and chlorobenzene were detected in groundwater in Clearview Landfill and City Park.

In addition, groundwater samples from deep monitoring wells MW-01D, MW-05D and MW-07D exhibited higher concentrations of several contaminants (e.g., metals) than those from the adjacent shallow wells. Groundwater samples collected from two monitoring wells (MW-10 and MW-11), located in the central part of Clearview Landfill exhibited high concentrations of the majority of contaminants. The groundwater samples collected in the Eastwick neighborhood had relatively low concentrations of contaminants.

The contaminants listed below were detected in groundwater samples at least once above their respective RBCs for tap water:

#### Metals

- |            |             |
|------------|-------------|
| • Aluminum | • Antimony  |
| • Arsenic  | • Barium    |
| • Cadmium  | • Copper    |
| • Iron     | • Manganese |
| • Nickel   | • Thallium  |
| • Vanadium | • Zinc      |

#### VOCs

- |                             |                                  |
|-----------------------------|----------------------------------|
| • 1,3-Dichlorobenzene       | • 1,4-Dichlorobenzene            |
| • 1,4-Dioxane (p-dioxane)   | • 2-Hexanone                     |
| • Benzene                   | • Carbon tetrachloride           |
| • Chloroform                | • Methyl tert-butyl ether (MTBE) |
| • Tetrachloroethylene (PCE) | • Trichloroethylene (TCE)        |
| • Vinyl chloride            |                                  |

#### SVOCs

- |                               |                              |
|-------------------------------|------------------------------|
| • 2,6-Dinitrotoluene          | • 2-Methylnaphthalene        |
| • 3,3-Dichlorobenzidine       | • 4,6-Dinitro-2-methylphenol |
| • 4-Nitroaniline              | • Atrazine                   |
| • Benzo(a)anthracene          | • Benzo(a)pyrene             |
| • Benzo(b)fluoranthene        | • Benzo(k) fluoranthene      |
| • Bis(2-chloroisopropyl)ether | • Bis(2-ethylhexyl)phthalate |
| • Carbazole                   | • Dibenzo(a,h)anthracene     |
| • Indeno(1,2,3-c,d)pyrene     | • Naphthalene                |
| • Nitrobenzene                | • N-nitroso-di-n-propylamine |

#### Pesticides/PCBs

- |                           |                           |
|---------------------------|---------------------------|
| • Aldrin                  | • Alpha BHC               |
| • Beta BHC                | • Dieldrin                |
| • Gamma BHC               | • Heptachlor              |
| • Heptachlor epoxide      | • P,P-DDD                 |
| • P,P-DDE                 | • P,P-DDT                 |
| • PCB-1016 (Aroclor 1016) | • PCB-1242 (Aroclor 1242) |
| • PCB-1248 (Aroclor 1248) | • PCB-1260 (Aroclor 1260) |

In general, if an analyte was detected in the groundwater samples, its concentration exceeded the RBCs for tap water at one or more locations. However, this trend was more common for metals in the groundwater samples from monitoring wells. Iron and manganese concentrations exceeded their RBCs in all monitoring wells except MW-06 and MW-12. Arsenic exceeded its RBC in seven monitoring wells (MW-01S, MW-02, MW-03, MW-07S, MW-09, MW-10, and MW-11). Cadmium exceeded its RBC in MW-10 and MW-11. Lead exceeded its RBC in five wells (MW-04, MW-05S, MW-10, MW-11, and

MW-12). MW-10 and MW-11 generally contained exceedances of RBCs for many contaminants that were not detected at other locations.

In summary, the groundwater contained landfill-related pollutants (e.g., metals, VOCs, SVOCs, pesticides and PCBs) in low to moderate concentrations that exceeded RBCs. Groundwater samples from monitoring wells MW-10 and MW-11 contained higher concentrations of most contaminants than any other groundwater sampling locations. Groundwater in the southern industrial area was also heavily contaminated.

Groundwater contamination within each zone is summarized below.

#### *Clearview Landfill*

**Metals** – Twelve metals were detected above RBCs in groundwater samples throughout Clearview Landfill, including aluminum, antimony, arsenic, barium, cadmium, copper, iron, manganese, nickel, thallium, vanadium, and zinc. Of these, arsenic, iron, manganese, and vanadium were more frequently detected at higher concentrations than others. As shown on Figure 4-40, high concentrations of arsenic were detected at two monitoring wells MW10 (46.4 µg/L) and MW11 (64.2 µg/L), as well as at five borings (GP215, 233, 237, 239, and 241, all in the southern industrial area), ranging from 10.1 to 201 µg/L. These locations were also detected with high concentrations of other aforementioned metals. The groundwater sample from soil boring GP241 contained the highest concentration of both arsenic (201 µg/L) and iron (547 mg/kg).

**SVOCs** – Many SVOCs were detected in groundwater throughout Clearview Landfill. The primary SVOCs detected in exceedance of RBCs were PAHs, including benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, and indeno(1,2,3-C,D)pyrene (Table 4-18). In Clearview Landfill, concentrations of benzo(a)pyrene, the primary risk driver among the detected PAH compounds, are shown on Figure 4-41 and ranged from non-detect up to 7.6 µg/L at GP218 (approximately 2,500 times the RBC of 0.003 µg/L) located near Darby Creek bank where leachate seeps were observed. Four boring locations (GP237, 238, 243, and 245) in the southern industrial area were also detected with benzo(a)pyrene at concentrations ranging from 2.8 to 7.1 µg/L.

**VOCs** – VOCs were detected in the landfill groundwater samples at concentrations exceeding their RBCs as listed in Table 4-16. However, VOCs were not detected as commonly in the groundwater samples as were other classes of contaminants. Fuel-related VOCs, such as benzene and MTBE, were more frequently detected in exceedance of their RBCs than solvent-related VOCs such as TCE. As shown on Figure 4-42, high concentrations of benzene were detected at several borings all in the southern industrial area (e.g., 14 µg/L at MW12 and 18 µg/L at GP243). Benzene was also detected in groundwater along the western bank near where leachate seeps were observed.

Similar to the distribution of benzene, high concentrations of chlorobenzene in groundwater samples were detected mostly in the southern industrial area. Groundwater samples from monitoring well MW12, and soil borings GP237 and 243 were detected with 130, 87, and 190 µg/L of benzene, respectively – all greater than its RBC of 90 µg/L.

**PCBs** – Two PCBs, PCB-1242 and PCB-1260, were commonly detected in the groundwater sampling locations within Clearview Landfill at concentrations exceeding the RBC (0.033 µg/L). As with the majority of the other organic contaminants, the highest total concentration of PCBs (54 µg/L) was detected within the southern industrial area, especially at soil boring GP243. The groundwater samples

from monitoring well MW11 and soil boring GP245 were also detected at total PCB concentrations of 9.7 and 7.4 µg/L, respectively.

**Pesticides** – Many pesticides were detected in groundwater throughout Clearview Landfill. High concentrations of total pesticides were detected mostly in the groundwater samples within the southern industrial area (e.g., 1.4 µg/L at GP215, 1.6 µg/L at GP233, and 8.2 µg/L at GP243). The groundwater sample from soil boring GP243 contained high concentrations of other contaminants such as benzo(a)pyrene, benzene, chlorobenzene, and PCBs. Concentrations of 10 pesticides exceeded their respective RBCs—aldrin, alpha BHC, beta BHC, dieldrin, heptachlor, heptachlor epoxide, lindane, p,p-DDD, p,p-DDE, and p,p-DDT.

#### *City Park*

**Metals** – Metals were detected above RBCs in the groundwater samples throughout the City Park area. Of these metals, the distribution of arsenic in groundwater is shown on Figure 4-40. Arsenic was detected at concentrations ranging from non-detect (only at four locations—GP019, GP046, GP048, and MW01D) up to 241 µg/L (at GP067, which was located east of the southern industrial area of the landfill). The second highest concentration was detected at GP-029, located in walking paths along the eastern edge of the landfill.

**SVOCs** – SVOCs were not commonly detected above RBCs in groundwater throughout the City Park area, except several PAHs. Benzo(a)pyrene, which was selected as representative of the detected PAHs, was detected only at four locations—GP033 (7 µg/L), GP050 (2 µg/L), GP067 (8 µg/L), and GP247 (2.9 µg/L). Soil boring GP033 was located in the northern open field part of City Park, and three other borings were located in the southern part of City Park.

**VOCs** – Six VOCs were detected within the City Park area at concentrations exceeding RBCs for tap water, including 1,4-dichlorobenzene, benzene, carbon tetrachloride, MTBE, TCE, and VC. VOCs were not detected at significant concentrations in groundwater within City Park, and less frequently than within the landfill. As seen on Figure 4-40, benzene as a fuel-related VOC was detected more often in exceedance of the RBC than solvent-related VOCs such as TCE. Seven locations (GP027, GP028, and GP031 through GP035) contained groundwater exceeding the RBC (0.34 µg/L), ranging from 2 to 7 µg/L. The highest concentration of benzene was detected at GP027 (7 µg/L) near the northern part of walking paths. Groundwater from GP034 near the southern industrial area of the landfill was also detected with 5 µg/L of benzene.

Chlorobenzene was also detected in the groundwater samples throughout City Park. Only two locations were detected at relatively high concentrations—70 µg/L at GP027 and 64 µg/L at GP035. However, both were below the RBC level (90 µg/L).

**PCBs** – PCBs were detected in groundwater at levels exceeding their respective RBCs within the City Park area. Elevated concentrations of PCBs were detected only in the northern and southern parts of City Park. The highest concentration of total PCBs was detected at GP037 near the southern industrial area of the landfill.

**Pesticides** – Pesticides were analyzed in groundwater samples collected throughout City Park. Total concentrations of pesticides ranged from non-detect to 1.67 µg/L. The highest concentration was detected within soil boring GP067, which was located adjacent to the southern industrial area of the landfill. A similar concentration was also detected at GP026 near the tennis courts.

### *Eastwick Neighborhood*

**Metals** – Several metals, including aluminum, arsenic, iron, manganese, thallium, and vanadium, were detected in the groundwater samples collected in the Eastwick neighborhood area. Of these, arsenic concentrations are shown on Figure 4-40. Arsenic was detected only in six locations at concentrations ranging from non-detect to 31.4 µg/L, mostly along the boundary of City Park. The highest concentration was detected within GP-017, located near the tennis courts in City Park.

**SVOCs** – As shown on Figure 4-41, no SVOCs were detected in groundwater at concentrations exceeding their RBCs within the Eastwick neighborhood area.

**VOCs** – VOCs were not detected in groundwater within the Eastwick neighborhood. Benzene was not detected in groundwater and chlorobenzene was detected only at one location GP10 (3 µg/L).

**PCBs** – No PCBs were detected within the Eastwick neighborhood area.

**Pesticides** – No pesticides were detected within the Eastwick neighborhood area.

### 4.5 Stormwater Evaluation

Based on site topography, seven distinct drainage basins were identified at Clearview Landfill, including Basins A through G, as shown on Figure 3-12. Each basin was evaluated for stormwater drainage and runoff. The stormwater samples were also collected and analyzed to determine its general quality. Appendix Q provides the complete stormwater sample analytical results.

The following tabular summary shows drainage basins with their potential sources of contamination detected in soil and groundwater within the basins:

<b>Basin</b>	<b>Area (acres)</b>	<b>Runoff Water Discharge Location</b>	<b>Possible Contaminant Sources within Basin</b>
<b>A</b>	22.52	<b>None</b> - Stormwater ponds onsite in swale southeast of industrial area – pond at center labeled Basin B; high recharge expected at ponded area	Drains entire southern industrial area – PCBs in shallow soil and significant groundwater contamination
<b>B</b>	0.19	<b>None</b> - Stormwater ponds onsite; perennially wet area	Perennially wet area at low point for Basin A – total PCBs detected in soil above 50,000 µg/kg
<b>C</b>	6.58	<b>None</b> - Stormwater ponds onsite in open area in north part of City Park	Northern City Park Area with significant soil and groundwater contamination
<b>D</b>	16.87	<b>Discharges to Darby Creek</b>	Other landfill area
<b>E</b>	2.28	<b>Discharges to Cobbs Creek</b>	Other landfill area
<b>F</b>	6.67	<b>Discharges to Cobbs Creek</b>	Other landfill area
<b>G</b>	11.21	<b>None</b> - Stormwater ponds onsite	Other landfill area



As Figure 3-12 shows, stormwater or runoff on Clearview Landfill drains either toward Darby and Cobbs Creeks, or inland toward the City Park. Four of these basins draining Clearview Landfill (i.e., Basins A, B, C, and G) are enclosed, and therefore do not drain into the creeks or the City of Philadelphia stormwater system. During the RI, the effects of these enclosed low areas were evident. Significant ponding of water occurs in Basins A, B, C, and G after high precipitation events. Note that some city storm sewer catch basins (sewer inlets) are shown to lie at the edges of these stormwater sub-basins; however, these catch basins in the Eastwick neighborhood or along the S. 84<sup>th</sup> Street bridge actually lie outside these drainage basins, and do not appear to drain any part of the landfill.

To evaluate runoff from Clearview Landfill, ponded stormwater samples during a storm event on May 22, 2003 were collected from the low areas of Basins A/B (water sample PD01), Basin C (water sample PD03) and Basin G (water sample PD02). Stormwater samples were also collected to evaluate the possibility of leachate entering sewer lines.

In addition, stormwater samples were collected from five outfall pipes (CS01 through CS05) during a dry period (on April 15, 2003) and during a storm event (on May 6, 2003), and from three ponding areas (PD01 through 03) during a wet period (on May 22, 2003). Figure 3-12 shows the locations of these samples. Leachate leaking from pipes in the bank of Cobbs Creek was observed near the sampling location CS02, but its connection to City of Philadelphia sewer pipes was unknown. The origins and purposes of these pipes were also unknown.

Tables 4-20 through 4-23 summarize the concentrations of contaminants detected in stormwater and ponded water samples, including VOCs, SVOCs, pesticide/PCBs, and TAL (total) metals, respectively. The RBCs for tap water were used for comparison purposes.

Of the aforementioned analytes, metals were most commonly detected in stormwater. However, only four metals were detected above either their RBCs for arsenic, iron, and manganese or the EPA action level for lead (15 µg/L). In general, metal concentrations in the stormwater samples collected during a storm event were higher than those collected during a dry period, except for the sample from CS02. This result suggested that more flushing of metals occurs during the storm event. Arsenic was detected above its RBC in all stormwater sampling locations, except one location, CS01 (near Cobbs Creek upstream of the landfill). Iron and manganese were also detected throughout these locations, but only one sample from CS03 in Basin C contained both metals above the RBCs. Lead was detected above the EPA Action level at CS04 near the S. 84<sup>th</sup> Street bridge. However, the sample from CS05 across Darby Creek was also detected with lead above the action level.

Three VOCs (1,4-dichlorobenzene, MTBE, and PCE) were detected above their RBCs in six stormwater samples. Of these, 1,4-dichlorobenzene was detected above the RBC in two locations (CS02 and CS03) during both dry and storm periods. MTBE was detected above the RBC only at two locations—CS01 (8 µg/L), which was located near Cobbs Creek upstream of the landfill; and CS05 (3 µg/L), which was located opposite side of the Darby Creek. Both samples were collected during a dry period. PCE was detected above the RBC only in sample CS03 during a dry period.

Five SVOCs, including benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, and indeno(1,2,3-C,D)pyrene were detected above the RBCs in the stormwater samples. However, these SVOCs were detected only at one location (CS04) on the south side of Clearview Landfill near 84<sup>th</sup> Street bridge during the storm event.

No PCBs were detected above the RBCs in the stormwater samples. Only one pesticide, heptachlor epoxide, was detected above its RBC at one location (CS02) during a dry period.

In the stormwater samples, the analytes whose concentrations exceed the RBC in at least one sample included:

#### Metals

- Arsenic
- Manganese
- Iron

#### SVOCs

- Benzo(a)anthracene
- Benzo(b)fluoranthene
- Indeno(1,2,3-c,d)pyrene
- Benzo(a)pyrene
- Benzo(k)fluoranthene

#### VOCs

- 1,4-Dichlorobenzene
- Tetrachloroethylene(PCE)
- MTBE (methyl tert-butyl ether)

### 4.6 Hydrogeologic Evaluation

The purpose of the hydrogeologic evaluation was to aid in understanding fate and migration pathways of groundwater contamination. The following information was used during this evaluation:

- Soil boring and monitoring well logs from boreholes drilled near Clearview Landfill.
- Regional geologic mapping presented on Figure 2-5.
- Groundwater level measurements; these were used to prepare a groundwater flow net on/near Clearview Landfill.
- Water level measurements at staff gages in the Darby and Cobbs Creeks, including tidally-influenced water levels (Figure 4-27).
- Topographic elevations presented on Figure 2-3.
- Historical aerial photographs presented on Figure 1-3.
- MODFLOW® groundwater modeling.
- Surface water drainage basins as presented on Figure 3-12.
- Visible creek-bank seeps shown on Figure 3-8.

#### 4.6.1 Clearview Landfill Stratigraphy

Based on soil boring and monitoring well drilling logs (Appendix B), seven geologic cross-sections of Clearview Landfill were generated to depict general stratigraphy of Clearview Landfill. Figures 4-46A through C show cross-sections near the southern industrial area (in west-to-east direction); northern part (in west-to-east direction), and central and western areas (in the north-to-south direction) of the landfill, respectively.

Clearview Landfill is situated on unconsolidated coastal plain sediments overlying bedrock of the Wissahickon Formation. Bedrock was encountered at depths ranging from approximately 18 feet bgs in the Eastwick neighborhood to as deep as approximately 44 feet bgs at monitoring well MW04 located in the southern industrial area of the landfill near the Darby Creek stream channel.

The geology and stratigraphy in each investigative zone of OU1 are described briefly below (from the surface layer downward):

#### *Clearview Landfill*

- Fill soil, concrete, and construction debris up to approximately 20 feet thick at ground surface, consistent with construction and miscellaneous waste disposal that occurred up to at least 2001 (visible on the 2002 aerial photography and during site visits for RI).
- Landfill wastes up to 75 feet thick at the highest point (center) of Clearview Landfill, including undifferentiated garbage, miscellaneous wastes, and fill soils.
- A discontinuous peat layer (an organic-rich marsh deposit) present in many boreholes, with a thickness of 1 to 3 feet.
- 10- to 15-foot thick layers (in total) of sand, gravel, silt, and clays, similar in description to natural Trenton Gravel deposits, but often difficult to distinguish from overlying fill soils.
- Wissahickon Formation rock, consisting of micaceous schist (with distinctive muscovite mica). The Wissahickon Formation typically showed a highly-weathered upper rock zone grading into harder, less-weathered deeper bedrock. Monitoring wells MW04, MW05D, and MW07D encountered Wissahickon Formation rock during drilling (see Figure 3-11 for locations).

#### *City Park*

- 1- to 2-foot thick (thicker in places) fill soil at ground surface. In some areas, particularly the northern open field of City Park, there is only a very thin soil fill cover, and wastes visibly protrude at ground surface. Aerial photographs indicate that the City Park area was largely covered with thick landfill waste and re-graded.
- Landfill wastes with 8 to 12 feet thickness (at least) in the formerly existed marshland below the current City Park area. Figure 1-3 shows the extent of marsh and stream channels, now filled with wastes and soil.
- A discontinuous natural organic peat layer (an organic-rich marsh deposit) with a thickness of 1 to 3 feet. This layer was present in many boreholes installed within the formerly existed marsh.
- Discontinuous sand, silts, and clays, with a total thickness of 10 to 15 feet. This layer lies above bedrock. These apparently natural soils are similar in description to natural Trenton Gravel deposits, but are often difficult to distinguish from overlying fill soils.
- Wissahickon Formation rock, consisting of micaceous schist (with distinctive muscovite mica). The Wissahickon Formation typically showed a highly-weathered upper rock zone grading into harder, less-weathered deeper bedrock. Monitoring wells MW1S and MW08 in the City Park encountered Wissahickon Formation rock.

#### *Eastwick Neighborhood*

- Re-worked fill soil and demolition debris in one-to-two feet thickness (thicker in places) at ground surface. Demolition debris is common (bricks, wood, etc.) and appears to be a relic of demolition of structures that pre-dated the current Eastwick townhouse construction in the mid-1970s.
- A 15- to 25-foot thick (in total) layer of discontinuous sand, silts, and clays. This layer lies above bedrock. These appear to be natural soils that are similar in description to natural Trenton Gravel deposits.
- Wissahickon Formation rock, consisting of micaceous schist (with distinctive muscovite mica). The Wissahickon Formation typically showed a highly-weathered upper rock zone grading into

harder, less-weathered deeper bedrock. This formation was encountered at boreholes GP007, GP008, GP010, GP011, GP013, GP014, and GP015 during drilling.

#### 4.6.2 Groundwater Flow Analysis

The groundwater hydrology was evaluated by several means: a flow net analysis, well pair analysis, a MODFLOW® groundwater model, and specific conductivity. Each method is discussed in the following sections.

##### 4.6.2.1 Flow Net Analysis

Figure 4-47 shows a flow net (groundwater elevation contours with perpendicular groundwater flow direction lines) constructed using groundwater elevations above the mean sea level (MSL) provided in Table 4-24. If a well pair was present, the water level in the shallow well was used to represent water table condition at the location.

##### *Groundwater Flow Directions*

Clearview Landfill is one of the highest points in the region. Based on the mapping of water levels in monitoring wells, the water level contour lines are shown on Figure 4-47. The groundwater mound (or high water table) exists under the landfill and produces radial groundwater flow away from Clearview Landfill. Regionally, groundwater flow is expected to be southwest toward the Delaware River. However, locally near Clearview Landfill, groundwater flows radially outward from the landfill toward the Darby and Cobbs Creeks, south below the southern industrial area, or east below the Eastwick neighborhood. Based on flow mapping presented on Figure 4-47, a significant portion of groundwater flows east under the Eastwick neighborhood.

##### *Groundwater Recharge Areas*

Groundwater recharge occurs throughout the Clearview Landfill area. Visually, little or no runoff occurs from Clearview Landfill during smaller storm events. Based on the water level mapping and groundwater flow directions presented on Figure 4-47, groundwater recharge occurs primarily in enclosed drainage basins (on the east side of the landfill) that do not drain into the Darby and Cobbs Creeks.

##### *Groundwater Discharge Areas*

Groundwater and/or leachate visibly discharged at seeps in the banks of the Darby and Cobbs Creeks north, west, and southwest of the landfill. Gas bubbles were observed in the base of the Darby and Cobbs Creeks, suggesting that groundwater/leachate seeps into the creek bed with actively decaying organic matter.

Leachate seeps were observed and mapped along much of the frontage of Clearview Landfill along the Darby and Cobbs Creeks (see Figure 3-8). The majority of seeps were distinguished as leachate-bearing by rust-colored discharge, strong odors, visual evidence of sheen/contamination, and the presence of visible debris/wastes protruding from the associated creek bank. However, the flow and persistence of these seeps was irregular and seasonal. Few or no seepage areas were observed during the drought year of 2003.

#### 4.6.2.2 Well Pair Analysis

Three pairs of shallow and deep wells (MW01S/D, MW05S/D, and MW07S/D) were constructed to evaluate vertical hydraulic gradients and vertical changes in the concentration of detected substances. These well pairs were located near the Angelo Place residences in City Park (MW01S/D); the western edge of the landfill near Darby Creek (MW05S/D); and the confluence of Darby and Cobbs Creeks north of the landfill (MW07S/D).

##### *Vertical Hydraulic Gradient*

Based on comparisons of water level elevation data in 2003 through 2006 (see Table 4-24), water table elevations in the shallow wells appeared to be higher than those in the deep wells, indicating downward hydraulic gradients at all three well pairs. Therefore, it is likely that hydraulically separate zones exist above/below discontinuous silt/clay layers. Groundwater below these silts/clay layers is semi-confined and not directly connected to the water-table aquifer in these areas. However, north and east of Clearview Landfill, fewer silt/clay confining layers exist below the water table, and semi-confined conditions may not exist in these areas.

##### *Concentration Gradient*

The concentrations of contaminants in groundwater within each well pair, including MW-01S/D, MW-05S/D, and MW-07S/D, were evaluated. The findings of this evaluation for each compound group are described briefly below:

- VOCs – At MW05S/D, several VOCs in low concentrations were detected in the shallow well, but no VOCs were detected in the deep well. At MW-07S/D, VOCs were detected in both wells, but all detected VOC concentrations were higher in the deeper well than in the shallow well. VOCs were not encountered within either well at well pair MW-01S/D.
- SVOCs – Several SVOCs were detected in the shallow wells of the well pairs MW01S/D and MW05S/D, but no SVOCs were detected in the deep wells at these locations. Concentrations of SVOCs below the RBC were detected in groundwater within the well pair MW-07S/D, with slightly higher concentrations within the deep well.
- Pesticides – In groundwater at well pair MW01S/D, only one pesticide, beta BHC, was detected below its RBC in the shallow well and no pesticides were detected in the deep well. At well pair MW05S/D, pesticides in groundwater were found only in the deeper well, mostly below the RBCs. At well pair MW07S/D, pesticides were detected in the groundwater of both shallow and deep wells; however, the pesticide concentrations in the deep well were slightly higher than those in the shallow well, but all below the RBCs.
- Metals – Many metals were detected at varying concentrations in groundwater at all well pairs. However, there was no distinct trend in their distribution. Therefore, it is difficult to determine any concentration patterns.

In summary, based on the concentration and water elevation data at the three well pair locations, the water table aquifer is underlain by a discontinuous semi-confined unit, separated by silt/clay varying in thickness from absent to at least 17 feet thick (at MW01S/D). In addition, it appears that the aquifer zones at MW07S/D are not connected, while shallow and deeper aquifers at MW01S/D and MW05S/D are slightly connected. While the shallow aquifers at the three locations were all under water table conditions

where landfill leachate could affect water quality, downward contaminant migration is not expected to be significant near the three well clusters.

#### 4.6.2.3 Groundwater Flow Modeling

To determine aquifer characteristics and predict groundwater seepage/discharge rates, groundwater flow conditions were further evaluated utilizing the Visual MODFLOW<sup>®</sup> software package. The full package of groundwater modeling is provided in Appendix R.

Groundwater modeling is a simplification of actual physical conditions and represents a useful approximation for predictive purposes. Therefore, modeling should always be accompanied by actual physical data for calibration (performed herein) and performance of sensitivity analyses. When properly calibrated and used to simulate a site, interpreted model findings can yield useful data regarding important hydrogeologic information, particularly such as the volume of water discharged to Darby/Cobbs Creeks under steady-state conditions.

##### *Groundwater Flow Modeling and Water Budget Analysis*

The groundwater mound in the center of Clearview Landfill provides a large amount of recharge to the groundwater system in the area. The permeability of the material is sufficiently high to allow this high level of infiltration, but low enough in the underlying strata to hold the water at such a high elevation, rather than allowing the groundwater table to equilibrate with the surrounding area. As a result, a groundwater mound about 20 feet high has formed within the man-made hill that comprises Clearview Landfill.

Similar to the findings from the flow net analysis, modeling indicates that groundwater flows radially away from the mound located in the central portion of Clearview Landfill. A large fraction of the flow from the landfill is directed east towards the Eastwick neighborhood. Substantial flow also exits into Cobbs Creek north of the landfill, and Darby Creek immediately west and south of the landfill near the S. 84<sup>th</sup> Street bridge where oily seepage has been observed in the east bank of Darby Creek. A zone of high groundwater flow (i.e., the waste-filled former channel) causes unusually low water levels below the City Park (MW02, MW03, and MW04). The model successfully simulated these phenomena.

The significant fraction of groundwater that flows under the Eastwick neighborhood is largely because of high recharge in the enclosed sub-basins (Figure 3-12), which normally do not discharge to any surface water body, but rather produce large groundwater recharge volumes on the east side of Clearview Landfill.

The City Park and part of the Eastwick neighborhood were originally a wetlands/marsh area. Hydrologically, this means that the Clearview Landfill was a regional groundwater discharge area, and groundwater flows regionally southeastward, but locally toward the Darby and Cobbs Creeks.

The development of Clearview Landfill and Eastwick neighborhood placed a layer of highly permeable waste and/or fill into the wetlands/marsh under water table conditions. Groundwater recharge, in the form of infiltration, occurs through the waste/fill. The infiltrating water leaches soluble materials, and also mobilizes particulates and some non-aqueous free-oil liquids from the waste if they exist. These free-oil liquids were encountered in several boreholes.

### *Model Calibration Results*

Anisotropic model calibration was performed to approximate the modeling results to the field data. The calibrated flow model indicated that increased recharge exists within the vicinity of the northern part of the City Park area and the southern industrial area of the landfill (corresponding to enclosed-low drainage Basin A). It also appears that recharge is high within the landfill itself, evidenced by the limited amount of runoff. Further, there appeared to be a groundwater drainage zone in the vicinity of monitoring well MW02, with a rapid flow direction change toward the southeast. In cross-section, the flow appeared to proceed downward from the highest portion of the mound, and outward away from the highest elevation. Flow appeared to move primarily horizontal, with some downward and upward components at various locations, depending on the interactions of the various hydraulic conductivities. The highest hydraulic conductivities were encountered in the Eastwick neighborhood residential community, while the lowest hydraulic conductivities were found near the central part of the landfill.

#### 4.6.2.4 Specific Conductivity Analysis

Figure 4-48 shows the extent of leachate migration into the groundwater system in the landfill area, based on specific conductivity measured in the groundwater samples from Geoprobe and monitoring wells. Specific conductivity is a measure of water's ability to conduct electricity and, therefore, found to be a good measure of total dissolved solids and salinity in groundwater - the higher the concentration of ionic (dissolved) constituents, the higher the conductivity. High specific conductivity in groundwater is usually an indicator of landfill leachate migration to groundwater. As shown on Figure 4-48, the "plume" of higher specific conductivity is largely located east of the limit of the landfill, and is consistent with an eastward component of groundwater flow shown on Figure 4-27. Therefore, it is likely that groundwater has carried dissolved constituents eastward below the Eastwick neighborhood.

Specific conductivity generally reflects the impacts of highly mobile, dissolved mineral salts such as chloride ions that are normally present in landfill leachate. These mobile salts are "conservative" tracers of groundwater contaminant migration that migrate faster in groundwater than most other organic and inorganic/metals contamination. Because most other organic and inorganic/metals contaminants migrate more slowly than these salts (an effect called "retardation"), the limits of groundwater-borne contamination are normally less extensive than these conservative leachate-related indicators (Freeze and Cherry, 1979).

Further, three irregularly shaped lobes of the specific conductivity plume shown on Figure 4-48 extend east/northeast from the limits of the landfill. These lobes correspond generally to zones of thicker sands, which would be expected to have higher hydraulic conductivity than other types of geology.

In contrast, the most southern part of the specific conductivity plume does not extend significantly beyond the limits of Clearview Landfill. This is probably a result of the following. First, the garbage-filled former channel of Darby Creek may act as a conduit for groundwater movement southwest into Darby Creek, thus diverting groundwater flow before it can migrate under the Eastwick neighborhood. Second, the soils below the southern part of the Eastwick neighborhood are more clayey and, therefore, expected to have a lower hydraulic conductivity. This combination of a high groundwater flow rate in a trash-filled former creek channel adjacent to a clayey soil zone inhibits groundwater movement below Eastwick in this area (near the south end of Buist Avenue), yet promotes southwest groundwater movement toward Darby Creek. This direction of groundwater flow would naturally discharge to the general area where leachate seepage was observed with an oily sheen and petroleum odors, leaking into Darby Creek just north of the 84<sup>th</sup> Street bridge.

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## 5.0 CONTAMINANT FATE AND TRANSPORT

In this section, the findings from the previous sections regarding site physical characteristics, source characteristics, and extent of contamination were combined and analyzed to predict contaminant fate and transport at Clearview Landfill.

The fate of organic and inorganic compounds placed in a landfill is largely determined by the natural attenuation processes occurring within the landfill. Contaminants tend to partition among aqueous, solid, and gaseous phases, and their mobility and fate are generally dependent on the magnitude of the preference for one phase relative to another, which is a function of the physical/chemical characteristics of both the contaminant and the phases present (Reinhart and Townsend, 1997).

Mechanisms of mobility and transformation include biotransformation, volatilization, dissolution and advection, sorption, and chemical reactions such as precipitation, reduction, oxidation, and hydrolysis. Biotransformation and chemical reactions can reduce contaminant mass; however, a more toxic and/or mobile compound may be produced (e.g., PCE to TCE or VC). Dissolution and advection result in the movement of the compound with the bulk flow through the refuse pore spaces. Similarly, volatilization and transportation by the product gas can remove volatile contaminants from the landfill. Sorption and precipitation can retard contaminant movement as the compounds interact with the solid phase. Transport can be influenced by compound complexation or chelation that can either retard movement if the complex becomes associated with the solid phase or enhance mobility if the compound “piggybacks” on a more soluble complexing agent (Reinhart and Townsend, 1997).

The primary removal mechanism for metals in unlined landfills appears to be washout. However, in cases where waste is mixed with large amount of soil (i.e., in Clearview Landfill), other mechanisms such as metal sulfide and hydroxide precipitation, and subsequent capture within the soil matrix by encapsulation, sorption, ion exchange, and filtration also play a significant role in attenuating metals. However, the potential for remobilization of precipitated/sorbed metals exists due to continuous infiltration and groundwater recharge.

### 5.1 Potential Routes of Migration

The release of contaminants from uncontrolled landfill sites occurs in all of three phases – liquid, solid, and gas. Therefore, the potential routes of contaminant migration at the Clearview Landfill include:

- Airborne migration of fugitive dust and landfill gas
- Soil/waste to groundwater migration
- Groundwater/leachate to surface water migration
- Surface runoff
- On-site sediment migration
- Creek sediment migration
- Subsurface vapor intrusion

### Airborne Migration

Airborne migration of fugitive dust could be a potential route of contaminant migration from Clearview Landfill. Although no dust sampling was conducted during the RI, this pathway is being evaluated as part of the HHRA.

Two key parameters controlling fugitive dust are (1) the wind, which erodes and transports material from the site, and (2) disturbance of the soil by human activities. The dust has the same contaminants as those sorbed to the soil. Use of the landfill generates dust from excavation, grading, and truck traffic. The amount of soil lost due to wind erosion at a site is a function of wind velocity, vegetative cover, soil/waste properties such as texture and moisture, and areal extent of exposed soil surface. Most of the surface area at Clearview Landfill is currently covered with natural vegetation, except some small bare soil spots in the central portion of the landfill. During the RI, dust in the air was observed infrequently, generally only on very dry and windy days.

By far the greatest source of fugitive dust would be remediation involving soil handling. Remediation of contaminated soil frequently involves excavation for treatment/disposal or containment using a cap. Excavation entails the following necessary operations, each of which generates large quantity of fugitive dust unless control measures are taken - transportation, dumping, storage in piles, conveyance to treatment unit, and treatment. Even if the soil does not require excavation, usually some form of surface grading is necessary and this generates fugitive dust.

Airborne migration of landfill gas is also a potential route of contaminant migration. At the Clearview Landfill, airborne migration of landfill gas (methane) does not appear to be a considerable release mechanism given the age of the landfill, minimal gas production as evidenced by the lack of methane in soil gas, and the likely washout of the majority of organic waste constituents due to no liner or cover systems; however, several soil gas sampling locations near the residential properties exhibited comparatively high methane content. Therefore, there is the potential for airborne migration of landfill gas.

Low level non-methane organic compounds (NMOCs) - a term to collectively refer to chemical compounds that participate in photochemical reactions, excluding methane - were detected in the subsurface. However, migration of these NMOCs via the airborne route are not likely, as the concentrations are low and the subsurface in the immediate vicinity of the landfill is saturated, thereby limiting their off-site migration.

### Soil/Waste-to-Groundwater Migration

The subsurface soil within a large area of Clearview Landfill has been impacted by a variety of wastes, primarily those related to municipal and construction waste. Analytical data indicate that the groundwater beneath Clearview Landfill has been impacted by site contaminants. Consequently, the soil/waste-to-groundwater pathway will continue to be a major transport pathway as long as contaminated soil and waste are in contact with groundwater, and precipitation continues to percolate into the vadose zone, transporting contaminants to groundwater.

Moreover, most of the waste within Clearview Landfill lies below the water table and is currently saturated. The groundwater (or leachate because it is in contact with the waste material) is situated directly over a fractured bedrock surface, further impacting deeper groundwater in the area. Significant

infiltration occurs due to the lack of an engineered landfill cover and drainage system, allowing mounding and retention of precipitation that has infiltrated through soil and waste, and impacted groundwater over the last 30+ years.

#### Groundwater/Leachate-to-Surface Water Migration

Analytical data indicated that groundwater underneath the landfill footprint has been impacted by site-related contaminants both horizontally and vertically. Clearview Landfill is underlain by an unconfined water table aquifer where groundwater flows from high elevations to low elevations by natural gradients. Therefore, contaminants in groundwater migrate in the general direction of groundwater flow toward Cobbs Creek north of the landfill, Darby Creek immediately west and south of the landfill, and the Eastwick neighborhood east of the landfill.

Since leachate seeps discharging to Darby Creek were observed along drainage swales located on the western and southern perimeters of the landfill, a pathway exists for the migration of leachate contaminants to surface water.

#### Surface Runoff

Surface soil samples collected from the landfill contained elevated levels of PAHs, PCBs, and metals. Surface runoff can pick up soil particles and transport them in overland flow for deposition at a lower land elevation or deliver sediment to receiving waters. As a result, surface water and sediment quality sampling in the creeks adjacent to the landfill has identified potential site-related contaminants, which may be the result of either surface runoff or the direct discharge of leachate (i.e., from the leachate seeps). In addition, stormwater runoff quality can deteriorate if waste material is exposed because of erosion. Consequently, surface runoff is considered a current and future migration pathway for landfill contaminants.

#### On-Site Sediment Migration

Most of the surface area in Clearview Landfill is currently well vegetated. However, the overall effectiveness of this vegetation is uncertain, as evidenced by erosion (e.g., rills and gullies), especially along the western side of the landfill along Darby Creek. Leachate seeps along the Darby/Cobbs Creeks banks nearest Clearview Landfill contain sediment. Therefore, on-site sediment from Clearview Landfill has the potential to migrate to downstream areas during periods of high precipitation or other events that can disturb sediment. Therefore, migration of sediment is a current and future migration pathway at the site.

#### Creek Sediment Migration

Contaminated sediments in the parts of Darby Creek nearest Clearview Landfill are subject to potential migration during periods of high stream flow. High stream flow could result in the entrainment and transport of contaminated sediments with surface water which drains into John Heinz NWR at Tinicum and ultimately reaches the Delaware River. Thus, migration of sediments in Darby Creek is a current and future migration pathway.

### Groundwater-to-Air Migration (Vapor Intrusion)

Due to the fact that the contaminated groundwater plume emanating from the landfill moves in the general direction of groundwater flow toward the Eastwick neighborhood where a residential community is located, concerns have been raised about the potential for the migration of vapors from volatile substances in the subsurface into overlying buildings.

Under certain conditions, vapor can intrude into homes through cracks in basement walls, drain tiles, sumps, foundations, and utility lines. The vapors may accumulate in dwellings or buildings to levels that may pose short-term safety hazards, acute health effects, or aesthetic problems. However, it has been confirmed that most of the homes in the Eastwick neighborhood have no basement. In addition, data collected during the soil gas survey performed in this neighborhood indicated that VOC concentrations in soil gas are not substantially high. Furthermore, a screening indoor air analysis (Johnson & Ettinger model) using soil gas data from vapor monitoring wells and soil borings installed within the City Park and Eastwick Neighborhood indicated that the calculated indoor air VOC concentrations were relatively insignificant. Therefore, this pathway of contaminant migration is unlikely.

A groundwater vapor intrusion model was conducted as part of the HHRA to determine if the vapor intrusion is a potential pathway of contaminants in groundwater to air. Groundwater at the site currently poses an unacceptable risk to human health because of many VOCs, including PCE, TCE, VC, and benzene, that were also detected in soil gas in this neighborhood. The migration pathway applies to existing homes within the historical landfill footprint and groundwater plume area, as well as any future homes constructed within the plume area.

### 5.2 Contaminant Persistence

Understanding the persistence characteristics of particular contaminants helps in determining the likelihood that a contaminant of concern will reach a point where people might be exposed (e.g., a private drinking water well).

Persistence is the ability of a biological or chemical contaminant to remain unchanged in composition, chemical state, and physical state over time (Dragun, 1988). In any given medium, persistence of chemicals largely depends on their structure and prevailing environmental conditions conducive to degradation or transformation (e.g., microbial population and nutrients available to support microbial growth).

This section compiles information taken from literature regarding the persistence of contaminants commonly encountered at the site as observed in various media during the RI. However, the information presented herein is not site-specific; therefore, should be used with caution for the purpose of analyzing the behavior or possible impacts of these chemicals at the site.

Based on findings from the RI, the contaminants of potential concern in various media at the Clearview Landfill site include PAHs, PCBs, pesticides, VOCs/SVOCs (other than PAHs and PCBs), and inorganics.

The persistence of each of these contaminant classes in the environment is generally described below:

## PAHs

PAHs have a strong affinity for soil and sediment; therefore, they have low mobility. PAHs are generally considered to be very persistent in the environment. Biodegradation of PAHs is likely, but is lengthy, and the overall persistence of individual PAHs varies with their molecular weight and other factors.

PAHs are biodegradable in soil systems. Lower molecular weight PAHs are transformed much more quickly than higher molecular weight PAHs (EPA, 1979). Lower molecular weight PAH components are more water soluble than higher molecular weight PAHs. Van Dueren and others (2002) reported that readily mobilized compounds, such as naphthalene, phenanthrene, and anthracene are slightly water soluble. Persistent PAHs, such as chrysene and benzo (a) pyrene, present even lower water solubilities. Pyrene and fluoranthene are exceptions because these compounds are more soluble than anthracene, but are not appreciably metabolized by soil microorganisms.

Other factors, such as insufficient bacterial membrane permeability, lack of enzyme specificity, and insufficient aerobic conditions, affect PAH persistence. PAHs may undergo significant interactions with soil organic matter. Intermediate PAH degradation products (metabolites) in soil treatment systems may also display toxicity. Complete mineralization of PAHs is slow and intermediates may remain for substantial periods of time.

Based on the light PAH/heavy PAH ratios calculated for the soil and sediment samples at the site, the majority of the PAHs associated with the site are the high molecular weight PAHs. Consequently, they can be considered very persistent in site soil and sediments.

Benzo(a)pyrene is moderately persistent in the environment. It readily binds to soils and does not leach to ground water, though it has been detected in some ground water. If released to water, it will adsorb very strongly to sediments and particulate matter. In most water and sediment environments, it will resist breakdown by microbes or reactive chemicals, but it may evaporate or be degraded by sunlight. Benzo(a)pyrene is expected to bioaccumulate in aquatic organisms that cannot metabolize it, including plankton, oysters and some fish.

## PCBs

The PCBs used in many industrial applications were chemical mixtures made up of a variety of individual chlorinated biphenyl components, known as congeners. Most commercial PCB mixtures are known in the United States by their industrial trade names. The most common trade name is Aroclor.

PCBs are considered to be very persistent organic chemicals and, therefore, may remain for long periods of time cycling between air, water, and soil. Biodegradation is the only process known to transform PCBs under environmental conditions, but only the lighter compounds are measurably biodegradable (USEPA, December 1979). Some fungi such as *Phanaerochaete chrysosporium* may biodegrade PCBs although such microorganisms may not exist in local soil. There is experimental evidence to suggest that heavier PCBs (five or more chlorines per molecule) can undergo photolytic degradation, but there are no data to suggest that this process operates under environmental conditions (EPA, December 1979). Base-, acid-, and neutral-promoted hydrolysis are considered to be inconsequential degradation mechanisms for PCBs (EPA, December 1979).

### Pesticides

Whether pesticides are sprayed, dusted, or applied directly to the soil, the soil is the ultimate sink. Bioaccumulation in the food chain is an important fate mechanism of pesticides. Photolysis of pesticides can break down some pesticides at a slow rate and is therefore a less important fate mechanism than bioaccumulation. Hydrolysis, oxidation, and photolysis are not generally important fate mechanisms for pesticides in soil or water. Hydrolysis half-lives are reported for several pesticides in periods of months to years. Volatilization may be an important loss mechanism for some pesticides (e.g., aldrin and dieldrin) in aquatic systems.

### VOCs

Monocyclic aromatic VOCs, such as benzene, toluene, ethylbenzene, and xylenes (BTEX), are not considered to be persistent contaminants in comparison to pesticides, PCBs, phthalate esters, and metals. Such compounds are subject to degradation via the action of both soil and aquatic microorganisms. However, the biodegradation of these compounds in the soil matrix is dependent on several factors, including the abundance of microflora, macronutrient availability, and other soil conditions (e.g., pH, temperature, and oxygen).

Although BTEX compounds are amenable to microbial degradation, the rate of degradation cannot be predicted without site-specific information on the availability of nutrients and the type of bacteria present. If these contaminants discharge to nearby streams, volatilization and biodegradation may occur relatively rapidly. For example, a first-order degradation rate constant for benzene was reported to be 0.11 day<sup>-1</sup> in aquatic systems (Lyman et al., 1990). This corresponds to an aquatic half-life of approximately 6 days.

Other monocyclic aromatics are subject to similar degradation processes in aquatic environments (EPA, 1982). However, chlorinated monocyclic aromatics such as 1,4-dichlorobenzene isomers and chlorobenzene are not expected to be as susceptible to microbial degradation. For example, a reported first-order biodegradation rate constant for chlorobenzene is 0.0045 day<sup>-1</sup> in aquatic systems (Lyman et al., 1990). This corresponds to a half-life of approximately 150 days.

### Inorganics

In general, metals cannot be degraded or readily detoxified. The chemistry of an individual metal depends on its physical and chemical properties, the associated waste matrix, and the soil. Significant downward transportation of metals from the soil surface occurs when the metal retention capacity of the soil is exceeded, or when metals are solubilized (e.g., at low pH). As the concentration of metals exceeds the ability of the soil to retain them, the metals will travel downward with the leaching waters (Van Dueren et al., 2002).

Persistence and behavior of inorganic contaminants commonly detected in various media at the site are discussed below:

**Arsenic** exists as either arsenate ( $\text{AsO}_4^{3-}$ ) or arsenite ( $\text{H}_2\text{AsO}_3^-$ ) in the subsurface environment. Of these, arsenite is the more toxic form of arsenic and arsenite compounds are reported to be 4-10 times more soluble than arsenate compounds. Arsenic is chemically very similar to phosphorus. Like phosphate, arsenate forms insoluble precipitates with iron, aluminum, and calcium. The adsorption of arsenite is strongly pH-dependent, with an increase in sorption of arsenite by kaolinite (a mineral consisting of aluminum silicate) and montmorillonite (a very soft silicate mineral that typically forms in microscopic

crystals, forming a clay) over a pH range of 3-9 units and a maximum adsorption by iron oxide at pH 7. The adsorption of arsenite has been found to be rapid and irreversible, and iron oxide, redox, and pH were the most important properties in controlling arsenite adsorption by soils (McLean and Bledsoe, 1992).

Both pH and the redox are important in assessing the fate of arsenic in soil. At high redox potentials, arsenate predominates and arsenic mobility is low. As the pH increases or the redox decreases, arsenite predominates. This reduced form of arsenic is more subject to leaching because of its high solubility. Formation of arsenite may also lead to the volatilization of arsine and methyl-arsines from soils. Under soil conditions of high organic matter, warm temperatures, adequate moisture, and other conditions conducive to microbial activity, the reaction sequence is driven towards methylation (a term used to denote the attachment or substitution of a methyl group on various substrates) and volatilization (McLean and Bledsoe, 1992).

Although the redox state of a system is important, arsenic solubility and transport is dominated by adsorption reactions that occur at the surface of reactive iron and aluminum oxide minerals. Adsorption of arsenic oxyanions by mineral surfaces is favored at low pH, and adsorption decreases in magnitude with increasing pH in a manner consistent with other anions (Sigg and Stumm, 1981). In general, arsenate is adsorbed to a greater extent than arsenite, except at elevated pH (>9) where the opposite occurs (Xu et al., 1988; Wilkie and Hering, 1996; Raven et al., 1998). Consequently, in most environmental systems arsenite is more mobile and bioavailable, hence more toxic than arsenate.

**Cadmium** may be adsorbed by clay minerals, carbonates, and iron and manganese oxides, or precipitated as cadmium carbonate, hydroxide, and phosphate. Evidence suggests that adsorption may be the primary mechanism of cadmium removal in soils. In soils and sediments contaminated with metal wastes, the greatest percentage of the total cadmium was associated with the exchangeable fraction. Cadmium concentrations have been shown to be limited by cadmium carbonate in neutral and alkaline soils. As with all cationic metals, the chemistry of cadmium in the soil environment is to a great extent controlled by pH. Under acidic conditions, cadmium solubility increases and very little adsorption of cadmium by soil colloids, hydrous oxides, and organic matter takes place. At pH values greater than 6, cadmium is adsorbed by the soil solid phase or is precipitated, and the solution concentrations of cadmium are greatly reduced. Cadmium forms soluble complexes with inorganic and organic ligands, in particular with chloride ions. The formation of these complexes will also increase cadmium mobility in soils (McLean and Bledsoe, 1992).

**Chromium** exists in two possible oxidation states: the trivalent chromium ( $\text{Cr}^{3+}$ ) and the hexavalent chromium ( $\text{Cr}^{6+}$ ). Hexavalent chromium ions are more toxic than trivalent chromium ions. Forms of hexavalent chromium in the environment are either chromate ion ( $\text{CrO}_4^{2-}$ ) or dichromate ion ( $\text{Cr}_2\text{O}_7^{2-}$ ), depending on pH. The dichromate ions pose a greater health hazard than chromate ions. Because of the anionic nature of hexavalent chromium, its association with soil surfaces is limited to positively charged exchange sites, the number of which decreases with increasing soil pH.

Iron and aluminum oxide surfaces will adsorb chromate ions at acidic and neutral pH ranges. The adsorption of hexavalent chromium by groundwater alluvium is primarily due to the iron oxides and hydroxides coating the alluvial particles. The adsorbed hexavalent chromium is, however, easily desorbed with groundwater recharge due to its nonspecific binding. The presence of chloride and nitrate has little effect on hexavalent chromium adsorption, whereas sulfate and phosphate inhibit adsorption. Hexavalent chromium is highly mobile in soils and is one of the metals that are highly mobile in alkaline soils. However, clay soil containing free iron and manganese oxides significantly retards hexavalent chromium mobility (McLean and Bledsoe, 1992).



Trivalent chromium forms hydroxy complexes in natural water, including  $\text{Cr}(\text{OH})_2^+$ ,  $\text{Cr}(\text{OH})^{2+}$ ,  $\text{Cr}(\text{OH})_3^0$ , and  $\text{Cr}(\text{OH})_4^-$ . Trivalent chromium is readily adsorbed by soils and is least mobile in soils at pH of 5 units. Hydroxy species of trivalent chromium precipitate at pH 4.5 units and complete precipitation of the hydroxy species occurs at pH 5.5 units. Hexavalent chromium can be reduced to trivalent chromium under normal soil pH and redox conditions in the presence of soil organic matter as the electron donor.

The reduction reaction proceeds at a slow rate at environmental pH and temperatures, and may require years in natural soils (McLean and Bledsoe, 1992).

**Copper** is retained in soils through exchange and specific adsorption mechanisms. At concentrations typically found in native soils, copper precipitates are not stable. This may not be the case in waste-soil systems (e.g., Clearview Landfill) and precipitation may be an important mechanism of retention. Copper is adsorbed to a greater extent by soils and soil constituents than most other metals, with the exception of lead. Copper, however, has a high affinity for soluble organic ligands and the formation of these complexes may greatly increase copper mobility in soils (McLean and Bledsoe, 1992).

**Iron and Manganese** are common elements in soil and widely distributed in nature. Iron exists in soil and minerals mainly as insoluble ferric oxide and iron sulfide (pyrite). Under reducing (anaerobic) conditions, the ferric iron ( $\text{Fe}^{3+}$ ) is reduced to ferrous iron ( $\text{Fe}^{2+}$ ) which is very soluble in water.

Manganese is a transitional element which exists in a variety of oxidation states. Manganese is widely distributed in nature, but does not occur as a free metal. Manganese exists in the soil principally as manganese dioxide, which is very insoluble in water. Under reducing conditions, the manganese in the dioxide form is reduced from an oxidation state of IV to II, and becomes soluble in water, as with ferric oxide.

Groundwater that contains appreciable amounts of iron or manganese is always devoid of dissolved oxygen (DO) and high in carbon dioxide. The high carbon dioxide content indicates that bacterial oxidation of organic matter has been extensive, and the absence of DO shows that anaerobic conditions have developed.

However, when a groundwater aquifer is recharged with oxygen-bearing water, it is sometimes noted that the soluble iron content in the water increases, which seems to contradict the above stated need for anaerobic conditions. The explanation is that the oxygen is consumed through the oxidation of insoluble pyrite ( $\text{FeS}_2$ ), leading to anaerobic conditions and the formation of soluble ferrous sulfate (Sawyer and McCarty, 1978).

**Lead** is a naturally occurring, bluish-gray metal that is found in small quantities in the earth's crust. Lead is an element forming approx. 0.002 % of the Earth's crust. The most important lead-bearing minerals are galena ( $\text{PbS}$ ), cerussite ( $\text{PbCO}_3$ ), crocoite ( $\text{PbCrO}_4$ ), and pyromorphite ( $\text{Pb}_5(\text{PO}_4)_3\text{Cl}$ ).

Lead is very stable and accumulates in the environment. Lead is present in a variety of organic and inorganic forms such as lead acetate, lead chloride, lead chromate, lead nitrate, and lead oxide. Pure lead is insoluble in water; however, the lead compounds vary in solubility from insoluble to water soluble.

Lead is primarily used in the manufacture of batteries, metal products, paints, and ceramic glazes. Most lead encountered in the environment today is inorganic (e.g., lead oxide and lead chloride). Organic lead compounds include a number of common high-pressure lubricants (lead soaps), and the gasoline anti-

knock agents such as tetraethyl lead (TEL) and tetramethyl lead (TML). TEL and TML are lipid-soluble liquids of high volatility and both are insoluble in water. Organic lead can be more toxic than inorganic lead because the body can readily absorb it.

Most human exposure to lead occurs through ingestion or inhalation. Uncontaminated soil contains lead concentrations less than 50 ppm, but soil lead levels in many urban areas exceed 200 ppm. (ATSDR, 2005). The EPA's standard for lead in bare soil in play areas is 400 mg/kg by weight and 1,200 mg/kg for non-play areas. This regulation applies to cleanup projects using federal funds. This value is for guidance only and is not enforceable.

Lead is adsorbed on small particles of dust in the air and, therefore, may be transported over a considerable area depending on wind speed, wind direction, precipitation and humidity. The majority is, however, removed from the air by precipitation, and deposited on vegetation and soil. Note that the incineration operation in the southern industrial area of the landfill was reported by a former site worker and the Delaware County incinerator was formerly located near the landfill.

Sediment also acts as an accumulation sink for lead compounds. Insoluble lead compounds (e.g., TEL/TML and lead oxide) sink and are adsorbed in the sediment or accumulate on suspended matter (in particular the clay fraction).

Groundwater is adversely affected by soluble lead compounds (e.g. lead chloride [up to 9.9 µg/L] and lead nitrate). Lead is not chemically affected by deoxygenated water.

The sorption rate depends on the properties of the soil. Lead has a considerable affinity with humic substances. The pH is important for the availability of lead from its compounds. As with other metals, a low pH is linked to a high degree of desorption into the soil solution. However, as lead is quite immobile, it remains in the soil and is not easily absorbed by plants. Therefore, soils represent an important sink for lead compounds.

**Mercury** is found in soil in several forms, including elemental mercury [ $\text{Hg}^0$ ], mercurous ions [ $\text{Hg}_2^{2+}$ ], and mercuric ions [ $\text{Hg}^{2+}$ ], depending on soil pH and redox potential. Both the mercurous and mercuric mercury ions are adsorbed by clay minerals, oxides, and organic matter. Adsorption is pH dependent, typically increasing with increasing pH. Mercurous and mercuric mercury are also immobilized by forming various precipitates. Mercurous mercury precipitates with chloride, phosphate, carbonate, and hydroxide; however, at concentrations of mercury commonly found in soils, only the phosphate precipitate is stable. In alkaline soils, mercuric mercury will precipitate with carbonate and hydroxide to form a stable solid phase. At lower pH and high chloride concentration, mercuric chloride ( $\text{HgCl}_2$ ) is formed. Divalent mercury will also form complexes with soluble organic matter, chlorides, and hydroxides that may contribute to its mobility (McLean and Bledsoe, 1992).

Under mildly reducing conditions, both organically bound mercury and inorganic mercury compounds may be degraded to the elemental form of mercury. Elemental mercury can readily be converted to methyl or ethyl mercury by biotic and abiotic processes. These are the most toxic forms of mercury. Removal of mercury from groundwater is generally through volatilization and/or precipitation rather than adsorption by clays, and increases with increased pH. The amount of mercury removed by volatilization appears to be affected by the solubility of the mercury compounds and soil adsorption capacity (McLean and Bledsoe, 1992).

**Phosphorus** is usually found as phosphate minerals in nature due to its high reactivity with other oxygen-containing compounds. Phosphate ions become chemisorbed (sorbed by chemical forces) on the surfaces of iron and aluminum minerals in acidic to neutral systems, and on calcium minerals in neutral to alkaline systems. In addition, phosphates can precipitate at practically all pH ranges by forming relatively insoluble phosphate compounds of aluminum, iron, and calcium. Therefore, both precipitation and chemisorption are involved in phosphorus retention in soils. Since chemisorption is involved, phosphorus retention can exceed the sorptive capacity of the soil. Movement of phosphorus through soil column is minimal until all sorption sites are occupied.

**Selenium** has received much attention in recent years since recent studies reported the high incidence of deformity and mortality of waterfowl resulting from the agricultural drainage water high in selenium. Such studies have led to a better understanding of the distribution and movement of selenium in soils and groundwater (McLean and Bledsoe, 1992).

Selenium exists in the soil environment in four oxidation states: selenide, elemental selenium, selenite, and selenate. The concentration and form of selenium in soils are governed by pH, redox condition, and soil composition. Selenate is the predominant form of selenium in calcareous soils, while selenite is the predominant form in acidic soil. Selenite binds to sesquioxides (especially iron oxides) and its removal increases with decreasing pH. Selenite adsorption decreases dramatically in the presence of phosphate and organic acids, but appears to be unaffected by the presence of sulfate or chloride. Precipitation is not a major mechanism of retention of selenite in soils (McLean and Bledsoe, 1992).

Selenate dominates under alkaline conditions. In contrast to selenite, selenate in soils is highly mobile. Selenate is adsorbed by amorphous iron oxide. Selenate seems to be adsorbed by weak exchange mechanisms similar to sulfate, in contrast to selenite that is specifically adsorbed by soils and soil constituents. No stable precipitates of selenate are expected to form under the pH and redox conditions of most soils (McLean and Bledsoe, 1992).

**Zinc** is readily adsorbed by clay minerals, carbonates, or hydrous oxides, with the large percent of total zinc being associated with iron and manganese oxides. Precipitation is not a major mechanism of retention of zinc in soils because of the relatively high solubility of zinc compounds. Precipitation may become a more important mechanism of zinc retention in soil-waste systems (i.e., Clearview Landfill). As with all cationic metals, zinc adsorption increases with pH. Zinc is known to hydrolyze at pH greater than 7.7 units, and these hydrolyzed species are strongly adsorbed to soil surface. Zinc forms complexes with inorganic and organic ligands that will affect its adsorption reactions with the soil surface (McLean and Bledsoe, 1992).

### 5.3 Contaminant Migration

This section discusses factors affecting contaminant migration for the media of concern, and evaluates some important natural processes governing the transport, transformation, and fate of contaminants in the subsurface environment.

#### 5.3.1 Factors Affecting Contaminant Migration

In the subsurface environment, contaminant transport is strongly influenced by groundwater flow and soil properties. Some factors are discussed briefly below.

### Hydraulic Conductivity

Hydraulic conductivity is a measure of the capability of a material to transmit water. The following table presents the range of expected values for the hydraulic conductivity of various geologic materials (LaGrega et al., 1994):

**Typical Ranges of Hydraulic Conductivity for Various Soils**

<b>Soil type</b>	<b>Hydraulic conductivity (cm/sec)</b>
Clean gravel	$1 \times 10^5$ to $1.0$
Clean sand or sand + gravel mixtures	$1.0$ to $1 \times 10^{-3}$
Fine sands and silts	$1 \times 10^{-2}$ to $1 \times 10^{-6}$
Silty clay and clay	$1 \times 10^{-5}$ to $1 \times 10^{-9}$
Municipal waste (saturated)*	$1 \times 10^{-3}$

\* USEPA, 1994

It was reported that Clearview Landfill received large amount of MSW. The hydraulic conductivity of MSW appears to be similar to that of clean sand or sand/gravel mixtures. The values in the table above provide a starting point for the assessment of groundwater flow. However, the values of hydraulic conductivity for a given material type may be quite different from that for the entire formation and these are generally estimated within an order of magnitude accuracy.

### Soil Type

In general, the mobility (transport) of contaminants, especially for metals, is strongly influenced by soil type (e.g., texture), the corresponding pH, and cation exchange capacity (CEC) of the soil. Clays are extremely important in adsorption reactions because of high CEC. In addition, soils high in humus or other organic matter also exhibit good exchange capacity. The type of soil mineral present is also an important factor. Many sorption reactions take place at the surface of iron and aluminum hydroxides and hydroxyl oxides. Therefore, the iron and aluminum content in soils is an essential factor governing the ability of soil for metal immobilization (USEPA, 1984).

Soil texture or particle size is another factor that can influence fixation of metals by soil. In general, finely textured soil immobilizes metals to a greater extent than coarsely textured soil. Finely textured soil also has a greater CEC (USEPA, 1984).

As described in Section 4.6.1 and drilling logs in Appendix B, layers of discontinued organic-rich peat, and/or sand/silt/clay exist in the subsurface throughout OU1. Therefore, these soil layers likely influenced the mobility of many inorganic contaminants.

### pH

Soil pH plays an important role in the retention of metals in soil and is a controlling factor in sorption/desorption and precipitation/solubilization reactions. In addition, the CEC of soil generally increases with an increase in pH. Even with a soil that has a high affinity for a specific metal, the degree to which the metal is fixed is a function of pH. Soil pH has been determined to be a major factor along with CEC for the fixation of zinc, molybdenum, mercury, and copper (USEPA, 1984). Although soil pH was not measured directly, it is assumed to be similar to pH in groundwater, which was in the neutral range.

### Oxidation-Reduction Potential

The oxidation-reduction or redox potential of a soil is very important in determining which species of an element is available for sorption, precipitation, or complexation. In general, the reduced forms of a metal are more soluble than the oxidized form. The redox potential of a soil system is usually altered through biological activity and a change in redox potential is correlated with changes in pH. Reducing conditions may be associated with a low pH, resulting in the formation of CO<sub>2</sub> and organic acids from the microbial degradation of organic matter. A reducing environment typically exists in saturated soils underneath the zones with high organic concentrations. The anaerobic conditions would enhance the mobility of metals. Iron is a good example of a metal which readily undergoes redox reactions. In the anaerobic or anoxic conditions, ferrous (Fe<sup>2+</sup>) which is a reduced form of iron predominates and is more soluble than ferric (Fe<sup>3+</sup>) (USEPA, 1984). The reducing condition persisted in the subsurface throughout OU1, as evidenced by negative ORP values in groundwater samples.

#### 5.3.2 Fate of Contaminants

Various natural processes that will affect the fate of contaminants in the subsurface are summarized in tabular form below:

#### **Summary of Natural Processes Affecting the Fate of Contaminants in the Subsurface**

<b>Process</b>	<b>Class of Chemical</b>	<b>Effect</b>
Sorption	Organic	Retardation
Precipitation	Inorganic	Retardation
Ion exchange	Inorganic	Retardation
Filtration	Organic/inorganic	Retardation
Chemical oxidation-reduction	Organic/inorganic	Transformation/retardation
Biological uptake	Organic/inorganic	Retardation
Biodegradation	Organic	Transformation
Hydrolysis	Organic	Transformation
Volatilization	Organic	Elimination by intermedia transfer
Dissolution	Organic/inorganic	Mobility enhancement
Ionization	Organic	Mobility enhancement
Complexation	Inorganic	Mobility enhancement
Immiscible phase	Organic	Various partitioning

Modified from LaGrega et al. (1994)

Retardation refers to processes that impede the transport of contaminants by removing or immobilizing them from a free state (i.e., aqueous solution or vapor). Prime examples of chemical processes that result in retardation are sorption and precipitation. It is important to note that the immobilized contaminants in the retardation processes are not transformed and the processes are reversible.

Attenuation refers to two types of processes: (1) irreversible removal and (2) transformation. Removal by an attenuation process differs from retardation in that it reduces the mass of a substance. A common example is a process that transfers the contaminant to another media (e.g., volatilization). The more common type of transformation is one that transforms the molecular structure of the substance (e.g., oxidation-reduction).

Some natural processes increase the mobility of substances in the subsurface. Examples include dissolution of organic substances and complexation of metallic ions. Such processes are categorized as mobility enhancement.

#### 5.3.2.1 Inorganics

Some factors that affect migration of inorganics are discussed below.

##### Precipitation

Precipitation is the converse of dissolution: the concentration of a solute exceeds the solubility of that particular compound, and any excess solute changes to a solid and thus precipitates out of the solution. This process is reversible. If the concentration of a solute drops below its solubility, dissolution of precipitate could occur. Precipitation is particularly applicable to heavy metals.

Precipitation depends greatly on pH of the soil/groundwater system. Most metals precipitates at high pH levels as hydroxides. However, continued elevation of pH will increase the solubility of amphoteric (having characteristics of both an acid and a base) metals such as nickel.

Precipitation of a given metal species also depends on redox potential (Eh). When used together with pH, an Eh/pH diagram can be developed for a given metal, which indicates the species that a given metal will exist as under various Eh and pH conditions.

Metal precipitation is also dependent on the presence of anions and competing cations that exceed the solubility product constant, which in turn causes the metal to form insoluble inorganic compounds. For example, based on the Eh/pH diagram for barium, one could assume that barium is soluble at all pHs and all Ehs. However, the presence of sulfate or phosphate ions will cause the barium to form insoluble barium sulfate or barium phosphate, and therefore be removed from solution and become immobile. Solubility product constants are indicators of the tendency of a given compound to form in solution on the basis of the concentrations of the individual cation and anion of a product.

##### Sorption/Ion Exchange

Ion exchange involves the sorption of ions in solution onto oppositely charged discrete sites on the surface of a soil particle. Ion exchange can be considered as a subcategory of sorption. Therefore, both terms, ion exchange and sorption, are used interchangeably to explain this phenomenon for inorganics. It is driven by attractive force of maintaining electrostatic neutrality: the electric charges of the surface of the soil are balanced by equivalent free ions of opposite charge. As such, a previously held ion of weaker affinity is exchanged by the soil for the ion in solution. For example, calcium ion in groundwater may be exchanged onto the clay surface, replacing existing sodium ions.

The capability of a soil to retain and exchange cations is quantified as the cation exchange capacity. Both organic and inorganic material surfaces are important for cation exchange capacity. Two important organic surfaces in nature are humic and fulvic acids. The general affinity of cations for humic acids as a whole decreases in the following order: trivalent cations > divalent cations > monovalent cations. Fulvic acids are water soluble, but cations such as  $\text{Fe}^{3+}$ ,  $\text{Al}^{3+}$ ,  $\text{Cr}^{3+}$ ,  $\text{Pb}^{2+}$ ,  $\text{Cu}^{2+}$ ,  $\text{Hg}^{2+}$ ,  $\text{Zn}^{2+}$ ,  $\text{Ni}^{2+}$ ,  $\text{Co}^{2+}$ ,  $\text{Cd}^{2+}$ , and  $\text{Mn}^{2+}$  can form insoluble complexes with fulvic acid.

However, soils frequently have far greater inorganic fraction than organic fraction. Of the inorganic soil particles, clay has a much higher cation exchange capacity than other inorganic particles because of its extremely large surface area, abounding with negatively charged sites. The exposed surface area of a unit mass of colloidal clay is at least 1,000 times that of coarse sand. However, it should be noted that the cation exchange capacity of clays varies considerably, depending on the clay mineralogy. The following table presents the cation exchange capacity of commonly encountered clay minerals (LaGrega et al., 1994):

Clay mineral	Cation exchange capacity (meq/100 g)
Kaolinite	3 – 15
Illite	10 – 40
Smectite (montmorillonite)	80 – 150
Attapulgite	20 – 30

In the process of ion exchange, ions compete for the exchange sites and displace a previously held cation, termed cation exchange selectivity. Typically displaced cations include sodium and calcium. Similar to the general affinity of cations to organic surfaces, multivalent cations are more strongly adsorbed than monovalent ions, and smaller cations tend to replace large cations. Cations are typically replaced in the following order:



However, this order is not absolute. For example, at high concentrations a cation with low replacing power such as  $\text{Na}^+$  can replace a cation higher in the series.

In addition, ion exchange capacity is strongly dependent on pH. As the pH is lowered below neutrality, hydrogen ions readily replace metal ions. A key point is that ion exchange of metals with soil may be partially reversible; saturated exchange sites can replace cations as concentrations of contaminants decline in groundwater and in response to pH changes. Further, the exchange capacity of a subsurface material can be saturated such that eventually the transport of contaminants is unaffected by ion exchange.

Soil particles can also possess positive surface charges. The edges of alumino-silicate clay minerals, or oxide surfaces, and organic matter can all possess positive charges, depending on the pH of the surrounding water. Positive surface charges will tend to accumulate anionic species such as phosphate, arsenate, molybdate, selenite, sulfate, borate, silicic acid, fluoride, halides, and nitrate.

Most soil surfaces, however, are predominately negatively charged. Typically, negatively charged constituent anions in solution are repulsed by the negatively charged surfaces. This repulsion leads to a process called anion exclusion which increases mobility for some compounds. Anionic species such as phosphate, arsenate, molybdate, selenite, sulfate, borate, silicic acid, fluoride, halides, and nitrate will therefore migrate faster through the soil because of the repulsion forces present.

#### 5.3.2.2 Organics

In addition to the general processes described above for inorganic contaminants, the mobility and persistence of organics contaminants are also influenced by decomposition and transformation resulting from microbial activities. Therefore, most of the factors affecting the microbial metabolism or soil microflora will affect the degradation of organics contaminants in soil and groundwater.

The sorption of organic constituents in soil may be the most important factor affecting the fate of organic compounds in soil system. Adsorption to soil constituents will affect the rate of volatilization, diffusion, and leaching, as well as the availability of the compounds to microbial degradation. The properties of a contaminant affect its sorption behavior significantly. These properties include (Piwoni and Keeley, 1990):

- Solubility
- Polar/ionic character
- Octanol/water partition coefficient ( $K_{ow}$ )
- Acid/base chemistry
- Oxidation/reduction chemistry

Organic contaminants can be divided into three subgroups to discuss their sorptive behavior: (1) ionic or charged species; (2) uncharged species; and (3) uncharged nonpolar species. In general, many of the common organic contaminants in groundwater are the nonpolar species, including trichloroethylene, tetrachloroethylene, the chlorinated benzenes, and the more soluble components of hydrocarbon fuels such as benzene, toluene, and xylene (BTEX). Other organic contaminants including pesticides and phenols exist in solution as either charged or polar molecules.

A simple rule of sorption might be “opposite attract” for charged species and “likes interact with likes” for uncharged species. In its natural state, soil is primarily composed of sand, silt, clay, water, and a highly variable amount of natural organic carbon. The latter profoundly complicates a soil’s sorptive properties. The combination of these characteristics describes the surfaces offered as sorptive sites to contaminants in water passing through the subsurface matrix. For example, silts and clays have much higher surface areas than sand, usually carry a negative charge, and almost invariably associate with natural organic matter (Piwoni and Keeley, 1990).

It can be deduced that sandy materials offer little in the way of sorptive surfaces to passing contaminants, while silts and clays, particularly those having substantial amounts of organic matter, provide a rich sorptive environment for all three categories of organic contaminants. Even the porous and highly productive aquifers which are composed of sands and gravels usually have a few percent of silts and clays, and can result in a substantial sorptive behavior (Piwoni and Keeley, 1990).

Under most contamination situations, the primary transporting means is water. One of the most important properties in the aqueous phase is pH, which determines the chemical form and, in turn, the mobility of all contaminants. As an example, pentachlorophenol (PCP) will primarily be an uncharged polar molecule in an aqueous solution at pH below 4.7, while it will become an anion at pH above this value, increasing its solubility from 14 to 90 mg/L. Other characteristics of water that can influence the behavior of contaminants include the salt content and the dissolved organic carbon content. Chlorides, for example, which are not usually of much concern when dealing with organic contaminants, can have an important effect on the mobility of various metals. Dissolved organic matter at relatively high concentrations in leachate has a significant effect on the mobility of most nonpolar organics (Piwoni and Keeley, 1990).

Most organic substances in the subsurface will undergo transformation to smaller molecules via oxidation and reduction mechanisms induced by the metabolic activity of native microorganisms. Such transformation is termed biodegradation. Due to the small amount of oxygen in the subsurface, most transformation occurs via reducing pathways of anaerobic processes. Anaerobic biodegradation occurs at very slow rate in the subsurface; however it favors dehalogenation of chlorinated compounds that



typically resist aerobic degradation. It is important to note that anaerobic degradation may not always transform organic compounds to less toxic or less mobile forms. For example, the anaerobic degradation of tetrachloroethylene (PCE) produces more toxic vinyl chloride (VC).

Hydrolysis is a chemical reaction in which water reacts with a compound to produce other compounds, involving the splitting of a bond, and the addition of the hydrogen ion and the hydroxide anion from the water. For most chemicals, hydrolysis has a relatively insignificant effect compared to other attenuation processes. However, for chlorinated compounds, which typically are not readily transformed by biodegradation, hydrolysis may play a significant role. Hydrolysis of chlorinated organics involves exchange of the hydroxyl group from a water molecule with an anionic group on a carbon atom. The reaction typically forms alcohols or alkenes. The rate of hydrolysis largely depends on the pH and temperature of the water.

### 5.3.3 Conceptual Migration Pathways

#### Soil and Waste Erosion

Both surface and subsurface soil samples collected from Clearview Landfill have shown high concentrations of organic and inorganic contaminants. Surface soil consists predominantly of rubble fill and other demolition wastes which were delivered from outside or the adjacent City Park area during the public recreational facility development. Since an engineered cover system was not implemented at Clearview Landfill, surface soil can be eroded and underlying waste can be exposed by stormwater during high precipitation events.

Some factors that need to be considered with respect to soil erosion include:

- Erosion on the west side of Clearview Landfill can transport contaminated soil and suspended sediment into Darby Creek. Erosion on the east side of Clearview Landfill can transport contaminants to enclosed low areas adjacent to the residential properties.
- Water pooled in low relief areas associated with enclosed drainage Basins A, B, C, and G (see Figure 3-12) infiltrates through the contaminated subsurface soil and waste, and migrates downward, transporting the contaminants off-site.
- Large quantities of debris and garbage are currently exposed throughout the surface of the landfill. These waste materials are dangerous for visitors/workers and may enter into the adjacent creeks. Both Cobbs and Darby Creeks near the landfill already have large quantities of debris (e.g., concrete debris, metal beams, cars, etc.) in the streams, posing a hazard for recreational visitors.

#### Groundwater Migration

Due to the high elevation of the landfill and its apparent high permeability, infiltration has led to creation of a mound of groundwater within the landfill, migrating radially off-site. This water flows towards Darby and Cobbs Creeks, south towards the southern industrial area, and east under the Eastwick neighborhood.

Some important migration factors that need to be considered with respect to groundwater migration include:

- Infiltration through the waste matrix within the landfill continues to occur and generates leachate. Leachate mixes with groundwater recharge and seeps radially outward into Darby and Cobbs Creeks as well as the Eastwick neighborhood.
- Groundwater mixed with leachate is the major source of seeps observed along the stream embankment of Darby and Cobbs Creeks near Clearview Landfill, and the oily discharge near the southern industrial area.
- Contaminated groundwater has migrated eastward beyond the current landfill footprint, as indicated by analytical results shown in Figures 4-40 through 45.

#### Soil Gas and Landfill Gas Migration

Landfill gas is generated as the organic fraction of waste decomposes. In addition, many compounds detected in soil gas samples result from volatilization of either free-product or VOCs in soil and groundwater. Numerous residential properties, including the townhomes on Angelo Place, were apparently built on top of the historical landfill footprint. Therefore, there are concerns regarding potential soil gas intrusion into these residences.

Soil gas containing VOCs was detected in soil borings installed throughout City Park and its differential pressure exerted was often sufficient to cause out-gassing from the soil borings. When natural soil venting to the atmosphere is obstructed by frozen or wet ground, the high pressure soil gas could carry soil vapor into residences through cracked floors or foundations (this potential migration pathway was not fully assessed yet.) However, there are no known anecdotal reports of actual problems resulting from soil gas accumulations or odors within the residential properties.

#### Leachate Migration

Leachate currently seeps from the banks of Clearview Landfill into Darby and Cobbs Creeks. The seeps observed on the banks were typically orange-stained, odorous, and sometimes showed an oily sheen. The results from flow net analysis and a groundwater flow model indicated that most groundwater beneath Clearview Landfill eventually seeps into Darby Creek (and Cobbs Creek to a lesser degree). Gas bubbles from the bed of Darby Creek near Clearview Landfill were frequently observed, apparently resulting from decaying organic matter present in the leachate seeping from the landfill.

The concentration of contaminants in leachate/groundwater seepage is diluted by surface water in Darby and Cobbs Creeks. The magnitude of this dilution depends on the degree of mixing and other factors such as tidal patterns, precipitation, etc.

Assuming seepage is completely mixed with surface water in the creeks, the magnitude of downstream dilution can be approximated by calculating a ratio of the combined annual flow in Darby and Cobbs Creeks to the seepage rate of leachate/groundwater determined by calibrated groundwater modeling, as follows:

Total Annual Flow in Darby and Cobbs Creeks	56,625 acre-ft/yr
Total Annual Leachate/Groundwater Seepage from Clearview Landfill	76 acre-ft/yr
Annualized Dilution Ratio	745:1

However, actual measured contaminant concentrations in downstream surface water were higher than those predicted by this dilution ratio. Consequently, background contamination in surface water is a significant contributor to downstream concentrations or seepage from the landfill does not mix completely with surface water in the creeks.

#### Stormwater Runoff

Stormwater runoff from Clearview Landfill flows either directly into Darby/Cobbs Creeks, or into enclosed low basins without external discharge under normal conditions (note that in 1999 Hurricane Floyd's extremely high floodwaters inundated most of the area surrounding Clearview Landfill.) In general, stormwater infiltrates in areas where soil permeability is high enough to allow water to penetrate the ground, and the slope is sufficiently level to prevent precipitation from running off. During infiltration, water passes through contaminated soil and waste, and generates leachate, contaminating underlying groundwater.

Only the drainage basins on the west and north sides of Clearview Landfill (Basins D, E and F in Figure 3-12) carry stormwater directly into the creeks. On the west side of Clearview Landfill, runoff has caused minor erosion due to steep slopes and/or impermeable soils.

On the east side of Clearview Landfill, several areas where stormwater runoff is collected do not drain to the stormwater system of the City of Philadelphia. These areas also have high groundwater recharge; therefore, the ground is saturated and frequently ponded. At this time, no drainage or access controls exist to prevent residents/visitors/workers from being exposed to runoff from these areas.

#### Creek Sediment and Surface Water Migration

Surface water in Darby and Cobbs Creeks near Clearview Landfill flows to the John Heinz NWR at Tinicum. Based on surface water and sediment transport and tidal analyses, together with visual inspection of surface water flow into Tinicum Marsh, the following conclusions reflect current conditions:

- Contaminants in the creeks tend to partition between solid and aqueous phases. Sediment is typically much more contaminated than surface water by the site-related contaminants such as PAHs. During periods of ebb tide and normal flow, surface water in Darby and Cobbs Creeks flows in the main channel of Darby Creek, not in the numerous channels of Tinicum Marsh. Therefore, downstream-flowing sediment potentially carrying contamination has a strong tendency to stay in the main channel of Darby Creek under these flow conditions.
- Normal flow (and ebb tide flow) carries more water downstream than flood-tide flows at a higher velocity (6 times more flow on average – see Appendix T); therefore, carrying larger volume (or grains) of sediment.
- Flow reversal during flood-tide conditions carries one-sixth as much water at a correspondingly lower velocity; therefore, transporting much finer suspended solids and bed-load sediment. Under flood-tide conditions, Tinicum Marsh is inundated during most tidal cycles and very fine-grained sediment is deposited on the marsh tidal flats.
- The impoundment at the NWR receives water from at least one tide gate along Darby Creek and does not normally connect with Darby Creek flow. Only under extreme flooding conditions does this impoundment receive overland flow from Darby Creek. Under this extreme condition,

contaminant transport from Clearview Landfill is possible, but only as part of a much larger load of contamination from many sources carried by extremely high (and correspondingly rare) floodwaters. Numerous other watershed-wide contaminants would be co-mingled with runoff/eroded sediment.

- Within Tinicum Marsh, there is extensive mixing of water and sediment during each tidal cycle, making it difficult to distinguish Clearview Landfill contaminants from those derived from other sources. The contaminants are very similar in nature, concentration, and migration properties.
- Historical occurrences of high-concentration contamination (such as PCBs from past Clearview Landfill operations) may have formerly existed in Darby Creek sediment, but have migrated through natural processes into the Delaware River/estuary.
- Combined sewer outfalls located upstream of Clearview Landfill carry discharges from sanitary wastewater from the City of Philadelphia and other non-point source runoff to Darby and Cobbs Creeks.

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## 6.0 RISK ASSESSMENT

A risk assessment was conducted to evaluate the potential risks to human health and the environment from release of contaminants at the LDCA site. The general objectives of the risk assessment were to estimate the actual or potential risks resulting from the presence of contamination attributable to the site and to provide the information for determining appropriate environmental cleanup measures for the media, if warranted. The specific goals of the baseline risk assessment were to:

- Identify and provide analysis of baseline risks (defined as risks that might exist if no additional remediation or institutional controls were applied at the site) and help determine what action is needed at the site.
- Provide a basis for determining the levels of chemicals that can remain on-site and still not adversely impact public health and the environment.
- Provide a basis for comparing potential impacts of various remedial alternatives.

The risk assessment results document the magnitude of potential risk at the site and associated cause(s) of that risk. The results may also be used to establish any remedial goal options that may be necessary. Finally, the results of the baseline risk assessment will help determine what, if any, remedial response actions may be necessary and assist in establishing clean-up goals.

Accordingly, the human health risk assessment (HHRA) and ecological risk assessment (ERA) were conducted, and the findings from these assessments are summarized herein. The full versions of the HHRA and ERA are provided in Appendices U and V, respectively.

### 6.1 Human Health Risk Assessment

Three major aspects of chemical contamination must be considered when assessing human health risks:

- Contaminants with toxic characteristics must be found in environmental media and must be released by either natural processes or by human action.
- Potential exposure points must exist either at the source or via migration pathways if exposure occurs at a remote location other than the source.
- Human or environmental receptors must be present at the point of exposure.

Risk is a function of both toxicity and exposure; without any one of the three factors listed above, there is no risk. The site-specific potential pathways for contaminant migration and exposure media which provide a potential route of contact with human receptors are illustrated in Figures 6-1 and 6-2.

To assess risks for these contaminant exposure pathways, the HHRA was divided into six components:

- Data evaluation (Section 6.1.1)
- Exposure assessment (Section 6.1.2)
- Toxicity assessment (Section 6.1.3)
- Risk characterization (Section 6.1.4)
- Uncertainty analysis (Section 6.1.5)

Each section is summarized below, with additional details and tables presented in Appendix U. The tables associated with the HHRA follow the format adopted by EPA RAGS, Volume I, Part D: Standardized Planning, Reporting, and Review of Superfund Risk Assessments (EPA, 2001a).

#### 6.1.1 Data Evaluation

Complete sample analytical results for all media are presented in Appendix U, Part 3. All analytical data used in the risk assessment were validated following USEPA Region III data validation procedures (USEPA, 1993b, 1994a). Before accepting data for use in the risk assessment, a data quality evaluation was performed. Based on this review, rejected or blank qualified data were not considered for use in the risk assessment, while estimated values were accepted for use given the indicated uncertainty. A discussion of analytical results that were excluded from the risk assessment and any other problems with the data, such as elevated detection limits, is presented in Appendix U (Part 1), Section 2.2. Prior to use, the data were adjusted to replace field duplicate pairs with the higher of the two concentrations and non-detected results with one-half of the sample detection limit or quantitation limit.

The sampled media used in the risk assessment include surface water, sediment, surface soil, subsurface soil, and groundwater. Concentrations in two exposure media were modeled to predict inter-media transfer, e.g., volatilization to air and fish tissue biouptake from sediment. The following notes apply to the risk assessment data sets representing areas of interest for each medium:

- The two creeks that flow towards and meet at a confluence alongside Clearview Landfill are Darby and Cobbs Creeks. Both creeks were sampled to include upstream and downstream locations, but upstream locations were removed from the data set used to evaluate possible site-related impacts. Since Cobbs and Darby Creeks represent a contiguous stream pathway, receptors could be exposed to both areas during the same exposure event. Therefore, sample analytical results for these creeks were combined into one data set to assess human health risks from all creek locations influenced by the Clearview Landfill.
- Two other surface water and sediment sample data groups were assessed: Tinicum Marsh, which is located downstream of the landfill, and is a receiving body for the surface water from Darby Creek and leachate seeps from the landfill.
- Surface soil data were subdivided into three groups due to different land use patterns in each area: Zone 1 surface soil samples were collected at Clearview Landfill and within the southern industrial area. Zone 2 surface soil samples were collected from the City Park adjacent to the landfill. Zone 3 samples were collected from the Eastwick residential neighborhood (see Figure 2-3). Surface soil samples represent a depth interval ranging from ground surface to two feet below surface.
- Subsurface soil data were combined into the same three geographic groups. Subsurface samples were collected from depth intervals two feet and deeper. In the risk assessment, the data sets for subsurface soil and surface soil were combined within each zone and have been denoted as “total soil” for the purpose of evaluating hypothetical future soil contact.

##### 6.1.1.1 Selection of Chemicals of Potential Concern (COPCs)

Analytical data from each exposure medium were screened against EPA Region 3 risk-based concentrations (RBCs) to determine which substances required quantitative calculation of risks. A substance was selected as a COPC if the maximum detected concentration in an area of interest exceeded its RBC that corresponds to an estimated lifetime cancer risk probability of one in a million ( $1 \times 10^{-6}$ ) or a



noncancer hazard quotient (HQ) of 0.1. Noncancer RBCs were adjusted to correspond to a HQ of 0.1, which is one-tenth of the usual HQ benchmark (1.0) used to assess single-chemical exposure, to ensure protectiveness against additive adverse effects for potential exposures involving multiple substances. RBC criteria for groundwater assume lifetime residential tap water use, while soil RBCs assume lifetime residential exposure to soil. RBCs for sediment and surface water applied a ten-fold multiplier to the corresponding soil and tap water residential RBCs due to lower frequency recreational exposures. Region 3 RBCs were also applied to select indoor and outdoor air COPCs using soil gas data, and for consumption of fish tissue. Appendix U (Part 1), Section 2.3 provides further details and limitations of the procedure used to select COPCs. The COPCs selected for each medium within area of interest are presented in Appendix U (Part 2) on RAGS D Table 2s, which list the maximum detected concentrations compared to RBCs.

#### 6.1.2 Exposure Assessment

The exposure assessment comprises a process for estimating chemical intakes for various receptors based on assumed typical quantities and rates of ingestion or contact with contaminated media, receptor-specific body measurements, and duration and frequency of exposure. Detailed equations and input parameters for each type of receptor and exposure medium are presented in Appendix U (Part 2) on RAGS D Table 4s. Details regarding receptor exposure parameters are given in Appendix U, Part 1, in Section 4.3. Modeled pathways of exposure that involve inter-media transfer, such as dermal absorption, volatilization to ambient air, and vapor intrusion are discussed in Appendix U, Part 1, Section 3.2.1, with parameter calculations shown in tables in Appendix U, parts 6, 9, 10, 11, and 12.

In the HHRA, potential human exposure routes that were assessed include tap water use of potable groundwater by future residents; direct contact with groundwater in excavations by industrial and construction workers; contact with soil by residents, recreational persons, construction workers, and industrial workers; contact with surface water and sediment by recreational persons and construction workers; ingestion of contaminated fish by recreational persons or subsistence fishers; inhalation of outdoor vapors emitted from soil gas by residents, recreational persons, construction workers, and industrial workers; inhalation of outdoor vapors emitted from groundwater in open excavations by construction workers and industrial workers; and inhalation of indoor air impacted by vapor intrusion of soil gas into residential dwellings and industrial buildings. The list of applicable receptors varied somewhat for the different areas of interest, as explained in Appendix U, Part 1, Section 4.3.

Sample concentrations for the COPCs in each data set were utilized collectively to estimate a typical value for the upper range concentrations to which a receptor may be continuously exposed while at or near the site. The estimation of exposure point concentrations (EPCs) provides a statistical procedure for estimating the chemical input into each of the exposure pathways. The 95 percent upper confidence limit (UCL) on the mean concentration was considered the input concentration of a chemical used to estimate site-associated risks for larger data sets (greater than 5 samples). For each substance, the 95 percent UCL was calculated using the methods presented in the EPA-approved software program, ProUCL version 3 (USEPA 2002c, Singh, 2004). If there were less than 5 samples in a data set, then the maximum detected concentration was selected as the EPC. EPCs are presented in Appendix U, Part 2 in RAGS D Table 3s.

Two types of exposure assumptions are possible for use in the HHRA: reasonable maximum exposure (RME) and central tendency exposure (CTE). RME is the exposure that is expected to represent a high end, but not usually worst-case, exposure in a given medium of concern. CTE is the exposure that is expected to represent a more typical exposure to receptors in a given medium of concern. The input

parameters associated with receptor activity patterns and other modeled variables were adjusted to represent central estimates. However, EPCs were assumed to be identical for RME and CTE evaluations.

Exposures to lead in environmental media were evaluated for residential children using EPA's Integrated Exposure Uptake Biokinetic (IEUBK) model (EPA, 1994b) which is a physiologically-based pharmacokinetic (PBPK) model. Exposures to lead by adult workers were assessed using the EPA Adult Lead Model (EPA, 2003b). These models predict blood lead levels in a population of lead-exposed children or in the fetus of an adult worker, based on lead biokinetic calculations which predict quasi-steady state blood lead concentrations in individuals who have relatively steady patterns of lead exposure. Details of the assumptions utilized in lead modeling are discussed in Appendix U, Part 1, in Section 4.4.

### 6.1.3 Toxicity Assessment

The toxicity assessment identifies the potential health hazards associated with exposure to each COPC. As discussed in Appendix U, Part 1, Section 5, dose-response values [non-cancer reference doses (RfDs) and cancer slope factors (SFs)] have been developed by EPA and other sources for many organics and inorganics. Non-cancer RfDs and cancer SFs associated with oral exposure, and corresponding toxicity values for inhalation exposure were obtained from the hierarchy of sources recommended by EPA. Non-cancer toxicity factors are presented in Appendix U, Part 2, in RAGS D Tables 5.1 and 5.2, and include RfDs associated with chronic/subchronic effects to particular target organs. Cancer SFs are presented in Appendix U, Part 2, in RAGS D Tables 6.1 and 6.2. Dermal exposure SFs and RfDs were based on extrapolation from oral toxicity values in accordance with EPA RAGS Part E Final Guidance for Dermal Exposure Assessment, while inhalation pathway toxicity values were obtained from independent studies in accordance with EPA RAGS Part F Final Guidance for Inhalation Exposure Assessment. Appendix U, Part 1, Section 5 also discusses chemical-specific toxicity assumptions for chromium, mercury, PAHs, dioxins/furans, vinyl chloride, and use of toxicity values for chemically similar surrogate compounds for compounds lacking published toxicity values. Lead is regulated by USEPA based on blood-lead uptake using IEUBK model. Based on residential exposures, lead is screened at 400 mg/kg in soil and sediment, and 15 µg/L for groundwater and surface water.

### 6.1.4 Risk Characterization

This section presents estimates of carcinogenic risks, noncarcinogenic hazards, and lead risks for applicable human receptors that are potentially exposed to COPCs identified in each medium within area of interest. The metric for evaluating cancer risks is based on estimating the probability of occurrence of cancer in an exposed population. The criteria for evaluating non-cancer hazards are based on the HQ, which is a unitless number indicating the ratio of estimated dose versus a published threshold dose representing a level above which adverse effects can no longer be ruled out. Lead risks are judged based on estimated blood lead concentrations, expressed in µg/dL, which are predicted for a certain percentage of individuals in an exposed population.

EPA has defined acceptable risks for the sum of cancer risks from all carcinogens as within the range of  $10^{-6}$  and  $10^{-4}$  excess lifetime cancer risk. For non-carcinogens, the benchmark level for acceptable risk is a HI of less than or equal to 1.0, which represents the sum of the HQs for all compounds affecting the same target organ. For lead, the benchmark level for acceptable risk is a blood lead concentration of 10 µg/dL predicted to be exceeded in no greater than 5 percent of an exposed population.

The following summary of human health risks is presented for each exposure area of interest and receptor. Appendix U provides a detailed discussion of chemical-specific risks along with supporting

documentation, which includes RAGS D Table 9s (listing risks for all COPCs). Appendix U, Table 10s present the risks for a shortened list of contaminants of concern (COCs) developed for each area of interest, which include only substances that contribute significantly to human health risks above the target acceptable risk range. A list of COC Risk drivers was not applicable if the medium-wide risk was less than the benchmarks discussed earlier. For noncarcinogenic chemicals, the list of COCs included those for which the HQ was greater than 0.2 and, when added to the HQs for other substances affecting the same target organ, yielded a HI value greater than 1.0. For carcinogenic chemicals, candidate COCs included all substances exhibiting an ICR greater than  $1.0 \times 10^{-6}$  within the particular exposure pathway. Lead was considered a COC when the model results exceeded the above benchmarks.

Table 6-1 summarizes the total risks of potential receptors exposed to media of concern within the area of interest and lists COCs contributing to the risks. COCs in several media were statistically demonstrated to be present at concentrations similar to background. Risk drivers that were similar to background were considered to be COCs, but with a caveat that the remedial planning process may consider eliminating these substances from the cleanup focus.

Figures 6-3, and 6-4a and 6-4b for soil and groundwater, respectively, provide the sample concentration results of risk driver substances on location tag maps. These figures show only the COCs that are cancer risk drivers or non-cancer risk drivers. Chemical concentrations are listed for those locations where a contributing chemical's individual cancer risk is greater than  $10^{-6}$  or where the HQ is greater than 0.2 and is a contributor to the target organ HI exceeding 1.0. Other documentation included in Appendix U includes Table 7s and Table 8s, which list the non-cancer and cancer receptor intakes, toxicity values, and EPCs, respectively; and estimated risks for each receptor and exposure medium. Appendix U, Part 7 contains the child lead model (IEUBK) prediction results and Appendix U, Part 8 contains the adult lead model (ALM) prediction results.

#### 6.1.4.1 Surface Soil Risks

##### Clearview Landfill

Exposure to Zone 1 (landfill area) surface soil was associated with estimated cumulative cancer risks that exceeded the acceptable risk range for lifetime residents ( $6.7\text{E-}4$ ) and lifetime recreational persons ( $1.3\text{E-}4$ ). For the recreational receptor, exposures would be more likely to occur over a shorter duration, so it should be noted that the recreational child cancer risk was  $9.0\text{E-}5$  and the recreational adult cancer risk was  $3.9\text{E-}5$ . COCs that contributed to these cancer risks included arsenic (which was similar to background based on statistical tests in Appendix U, Part 4), PCBs, dieldrin, heptachlor epoxide, benz(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, dibenz(a,h)anthracene, and indeno(1,2,3-cd)pyrene.

For the child resident, adult resident, and recreational child exposed to Zone 1 surface soil, the maximum of each receptor's estimated target organ HIs were 19, 2.4, and 3.2, respectively, which exceeded the benchmark HI of 1.0. The maximum estimated target organ HI was less than 1 for the recreational adult. Antimony was the largest contributor (HQ of 15.9 for the residential child) to unacceptable noncancer HIs. Other non-cancer risk drivers contributing to the unacceptable estimated target organ HIs include heptachlor epoxide, dieldrin, endrin, Aroclor-1254, cobalt, iron, thallium, and copper. The latter three metals were demonstrated to be similar to background based on statistical tests in Appendix U, Part 4.

Risks associated with exposure to a mean lead concentration of 372 mg/kg in Zone 1 surface soil were evaluated for an adult recreational receptor using the ALM model and for a residential child using the

IEUBK model. (Lead risks were not appropriate to estimate using either model for an adult resident or a child recreational receptor.) A predicted blood lead level above 10 µg/dL in less than 5 percent of the receptor population is considered protective. Blood lead concentrations were predicted to exceed 10 µg/dL in 6.3 percent of an exposed population of child residents, which indicates that adverse effects cannot be ruled out. For adult recreational receptors, less than 5 percent of an exposed population was predicted to have blood lead concentrations above 10 µg/dL. Soil lead concentrations exceeded background.

### City Park

Exposure to Zone 2 (the City Park area) surface soil was associated with estimated cumulative cancer risks that exceeded the acceptable risk range for residents (lifetime risk of 2.8E-4) but not for recreational persons (lifetime risk of 5.5E-5). For the lifetime resident, the COCs that contributed to cancer risks included arsenic (which was similar to background), Aroclors-1260 and 1254, benz(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, dibenz(a,h)anthracene, and indeno(1,2,3-cd)pyrene.

For the child resident exposed to Zone 2 surface soil, the maximum of the estimated target organ HIs was 2.6, which exceeded the benchmark HI level of 1.0. The maximum estimated target organ HI was less than 1 for the residential adult, recreational child, and recreational adult. For the residential child, Aroclor-1254 (HQ of 1.7) was the largest contributor to unacceptable noncancer HIs, with additional contributions from cobalt and iron, which were similar to background, as well as antimony.

Risks associated with exposure to a mean lead concentration of 282 mg/kg in Zone 2 surface soil were evaluated for an adult recreational receptor using the ALM model and for a residential child using the IEUBK model. (Lead risks were not appropriate to estimate using either model for an adult resident or a child recreational receptor.) Blood lead concentrations were not predicted to exceed 10 µg/dL for either receptor exposed to Zone 2 surface soil.

### Eastwick Neighborhood

Exposure to Zone 3 (the Eastwick Neighborhood area) surface soil was associated with estimated cumulative cancer risks that exceeded the acceptable risk range for residents (lifetime risk of 1.9E-4), but not for industrial worker (cancer risk of 1.2E-5). For the lifetime resident, the COCs that significantly contributed to cancer risks included arsenic (which was similar to background), benz(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, dibenz(a,h)anthracene, indeno(1,2,3-cd)pyrene, Aroclor-1260, and chloroform (for vapor intrusion from soil gas to indoor air).

The maximum of the target organ-specific HIs exceeded 1.0 for the residential child (maximum HI of 1.2), but HIs were acceptable for the residential adult and industrial worker. For the residential child, the significant contributors to unacceptable noncancer HIs included cobalt and iron, which were similar to background, and antimony.

Lead was not selected as a COC for Zone 3 surface soil.

#### 6.1.4.2 Total Soil Risks

##### Clearview Landfill

Exposure to Zone 1 total soil (representing combined surface and subsurface soil) was associated with estimated cumulative cancer risks that exceeded the acceptable risk range for an industrial worker ( $3.9\text{E}-4$ ). Cancer risks for the construction worker were within the acceptable risk range ( $5.2\text{E}-5$ ). The COCs that contributed significantly to cancer risks included arsenic (which was similar to background), Aroclors-1260, 1254, and 1268, dieldrin, heptachlor epoxide, 2,3,7,8-TCDD toxicity equivalents, benz(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, dibenz(a,h)anthracene, and indeno(1,2,3-cd)pyrene.

For the industrial and construction workers exposed to Zone 1 total soil, the maximum of each receptor's estimated target organ HIs were 1.2 and 5.2, respectively, which exceeded the benchmark HI of 1.0. Antimony was the largest contributor to unacceptable noncancer HIs (HQ of 4.5 for the construction worker), with a smaller contribution from heptachlor epoxide.

Risks associated with exposure to a mean lead concentration of 372 mg/kg in Zone 1 total soil were evaluated for a construction worker and an industrial worker using the ALM model. Blood lead concentrations were predicted to exceed 10  $\mu\text{g}/\text{dL}$  in 19 percent of an exposed population of construction workers, which indicates that adverse effects cannot be ruled out. For industrial workers, less than 5 percent of an exposed population was predicted to have blood lead concentrations above 10  $\mu\text{g}/\text{dL}$ . Soil lead concentrations exceeded background.

##### City Park

Exposure to Zone 2 total soil was associated with estimated cumulative cancer risks that exceeded the acceptable risk range for residents (lifetime risk of  $2.3\text{E}-4$ ), but not for the recreational persons (lifetime risk of  $4.6\text{E}-5$ ), industrial worker (cancer risk of  $1.6\text{E}-5$ ), or construction worker (cancer risk of  $2.3\text{E}-6$ ). For the lifetime resident, the COCs that significantly contributed to cancer risks include arsenic (which was similar to background), Aroclors-1260 and 1254, benz(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, dibenz(a,h)anthracene, and indeno(1,2,3-cd)pyrene.

For the child resident exposed to Zone 2 total soil, the maximum of the estimated target organ HIs was 1.2, which exceeded the benchmark HI level of 1.0. The maximum estimated target organ HI was less than 1 for the residential adult, recreational child and adult, industrial worker, and construction worker. For the residential child, Aroclor-1254 (HQ of 0.4) was the largest contributor to unacceptable noncancer HIs, with additional contributions from cobalt and iron, which were similar to background, and antimony.

For the construction worker exposed to a mean lead concentration of 227 mg/kg in Zone 2 total soil, the predicted percentage of a hypothetical population of pregnant workers that would exhibit a fetal blood lead level above 10  $\mu\text{g}/\text{dL}$  was 9.3%, which exceeds the protective level cutoff set at 5 percent of an exposed population. Blood lead concentrations were not predicted to exceed 10  $\mu\text{g}/\text{dL}$  for the child resident, adult recreational person, or industrial worker exposed to Zone 2 total soil. Soil lead concentrations were statistically demonstrated to be similar to background.

### Eastwick Neighborhood

Exposure to Zone 3 (the Eastwick neighborhood area) total soil was associated with estimated cumulative cancer risks that exceeded the acceptable risk range for residents (lifetime risk of  $1.3\text{E-}4$ ), but not for the industrial workers (cancer risk of  $8.6\text{E-}6$ ) or construction workers (cancer risk of  $1.2\text{E-}6$ ). For the lifetime resident, the COCs that significantly contributed to cancer risks included arsenic (which was similar to background), benz(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, dibenz(a,h)anthracene, indeno(1,2,3-cd)pyrene, and chloroform (for vapor intrusion from soil gas to indoor air).

Noncancer HIs did not exceed 1.0 for residential receptors, the industrial worker, or the construction worker. Lead was not selected as a COC for Zone 3 total soil.

#### 6.1.4.3 Surface Water and Sediment Risks

### Darby and Cobbs Creeks

For exposure to surface water and sediment from Darby and Cobbs Creeks, the estimated cumulative cancer risks for recreational persons were within the acceptable risk range (lifetime risks of  $9.1\text{E-}6$  and  $2.1\text{E-}5$ , respectively). Cancer risks for the construction worker were less than  $1\text{E-}6$ , the lower end of the acceptable risk range. Non-cancer HIs for surface water and exposure to these creeks did not exceed 1.0 for the construction worker or the child and adult recreational receptors. Blood lead modeling was not applicable to surface water and sediment concentrations because the IEUBK and adult lead models have not been developed or calibrated for this type of exposure scenario.

### Tinicum Marsh

For exposure to surface water and sediment from Tinicum Marsh, the estimated cumulative cancer risks for recreational persons were within the acceptable risk range (lifetime risks of less than  $1\text{E-}6$  and  $8.4\text{E-}6$ , respectively). Cancer risks for the construction worker were less than  $1\text{E-}6$ , the lower end of the acceptable risk range. Noncancer HIs for surface water and sediment exposure to Tinicum Marsh did not exceed 1.0 for the construction worker or the child and adult recreational receptors. Blood lead modeling was not applicable to surface water and sediment concentrations because the IEUBK and adult lead models have not been developed or calibrated for this type of exposure scenario.

### Leachate Seeps

For exposure to surface water and sediment from leachate seeps, the estimated cumulative cancer risks for recreational persons were within the acceptable risk range (lifetime risks of less than  $1\text{E-}6$  and  $6.1\text{E-}6$ , respectively). Cancer risks for the construction worker were less than  $1\text{E-}6$ , the lower end of the acceptable risk range. Noncancer HIs for surface water and sediment exposure to leachate seeps did not exceed 1.0 for the construction worker or the child and adult recreational receptors. Blood lead modeling was not applicable to surface water and sediment concentrations because the IEUBK and adult lead models have not been developed or calibrated for this type of exposure scenario.

#### 6.1.4.4 Fish Tissue Risks

##### Darby and Cobbs Creeks

Cancer risks were estimated for the lifetime consumer that recreationally catches fish from Darby and Cobbs Creeks, and for the lifetime receptor that engages in subsistence fishing. As shown in Table 6-1, the estimated cumulative cancer risks exceeded  $1\text{E-}4$  for lifetime subsistence (cancer risk of  $3.6\text{E-}2$ ) and lifetime recreational fishers (cancer risk of  $5.3\text{E-}3$ ). For the recreational receptor, exposures would be more likely to occur over a shorter duration, so it should be noted that the recreational child cancer risk was  $2.7\text{E-}3$  and the recreational adult cancer risk was  $2.6\text{E-}3$ . The COCs that contributed significantly to cancer risks included arsenic, PCBs, DDE, DDT, alpha- and gamma-chlordane, beta-BHC, dieldrin, heptachlor, heptachlor epoxide, benz(a)anthracene, BAP, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, dibenz(a,h)anthracene, and indeno(1,2,3-cd)pyrene.

For the child and adult subsistence fisher, and child and adult recreational consumers of fish from Darby and Cobbs Creeks, the maximum of each receptor's estimated target organ HIs were 197, 47, 29, and 6.9, respectively, all of which exceed the benchmark HI level of 1.0. The COCs contributing significantly to non-cancer hazards for these receptors include arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, zinc, alpha- and gamma-chlordane, dieldrin, heptachlor epoxide, DDT, fluoranthene, phenanthrene, and pyrene.

Blood lead risks were estimated for consumption of fish tissue containing a mean lead concentration of 90.4 mg/kg in fish presumed to be found in Darby and Cobbs Creeks. A mean creek sediment concentration of 149 mg/kg was applied to estimate biouptake based on modeling equations presented in Appendix U, Part 2, in RAGS D Table 2s. Blood lead concentrations were predicted to exceed  $10\text{ }\mu\text{g/dL}$  in 100 percent of individuals exposed, including child and adult subsistence fishers and child and adult recreational consumers of fish. Adult and child consumers of fish were evaluated using the ALM and the IEUBK model, respectively. Adverse effects cannot be ruled out when the predicted blood lead level exceeds  $10\text{ }\mu\text{g/dL}$  more than 5 percent of the receptor population.

##### Tinicum Marsh

Cancer risks were estimated for the lifetime consumer that recreationally catches fish from Tinicum Marsh, and for the lifetime receptor that engages in subsistence fishing. As shown in Table 6-1, the estimated cumulative cancer risks exceeded the acceptable risk range ( $1\text{E-}4$ ) for lifetime subsistence fishers (cancer risk of  $3.7\text{E-}2$ ) and recreational fishers (cancer risk of  $5.4\text{E-}3$ ). For the recreational receptor, exposures would be more likely to occur over a shorter duration, so it should be noted that the recreational child cancer risk was  $2.8\text{E-}3$  and the recreational adult cancer risk was  $2.6\text{E-}3$ . The COCs that contributed significantly to cancer risks included arsenic, DDE, DDT, alpha- and gamma-chlordane, alpha-BHC, dieldrin, heptachlor epoxide, benz(a)anthracene, BAP, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, dibenz(a,h)anthracene, and indeno(1,2,3-cd)pyrene.

For the child and adult subsistence fisher and child and adult recreational consumers of fish from Tinicum Marsh, the maximum of each receptor's estimated target organ HIs were 319, 76, 47, and 11, respectively, all of which exceed the benchmark HI level of 1.0. The COCs contributing significantly to non-cancer hazards for these receptors include arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, zinc, alpha- and gamma-chlordane, dieldrin, heptachlor epoxide, and DDT.

Blood lead risks were estimated for consumption of fish tissue containing a mean lead concentration of 111.7 mg/kg in fish presumed to be found in Tinicum Marsh. A mean marsh sediment concentration of 184 mg/kg was applied to estimate biouptake based on modeling equations presented in Appendix U, Part 2, in RAGS D Table 2s. Blood lead concentrations were predicted to exceed 10 µg/dL in 100 percent of individuals exposed, including child and adult subsistence fishers, and child and adult recreational consumers of fish. Adult and child consumers of fish were evaluated using the ALM and the IEUBK model, respectively. Adverse effects cannot be ruled out when the predicted blood lead level exceeds 10 µg/dL more than 5 percent of the receptor population.

#### 6.1.4.5 Groundwater Risks

Exposure to groundwater was associated with estimated cumulative cancer risks that exceeded the acceptable risk range for lifetime residents (6.6E-2) and industrial workers (2.5E-4), but not for construction workers (1.0E-5). As summarized in Table 6-1, the major cancer risk drivers with individual cancer risks above 1.0E-5 for the lifetime resident exposed via tap water consumption were 2,3,7,8-TCDD TEQ, arsenic, Aroclors-1260, 1242, and 1248, atrazine, aldrin, dieldrin, heptachlor epoxide, 2,4-dinitrotoluene, 2,6-dinitrotoluene, 3,3'-dichlorobenzidine, benz(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, dibenz(a,h)anthracene, indeno(1,2,3-cd)pyrene, bis(2-ethylhexyl)phthalate, N-nitroso-di-n-propylamine, PCE, and vinyl chloride. Additional cancer risk drivers displaying risks between 1E-6 and 1E-5 were 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, alpha-BHC, beta-BHC, delta-BHC, gamma-BHC (Lindane), gamma-chlordane, heptachlor, Aroclor-1016, 2,2'-oxybis(1-chloropropane), 3-nitroaniline, 4-nitroaniline, benzene, carbon tetrachloride, chloroform, trichloroethene, naphthalene, and 1,4-dichlorobenzene. Collectively, groundwater exposure to the lifetime resident poses the highest risk among the exposure scenarios assessed for the site.

For the industrial worker exposed to groundwater (i.e., in an excavation trench), the major cancer risk drivers were 2,3,7,8-TCDD TEQs, Aroclor-1260, benzo(a)pyrene, dibenz(a,h)anthracene, benzo(b)fluoranthene, benzo(k)fluoranthene, indeno(1,2,3-cd)pyrene, benzo(a)anthracene, naphthalene, chloroform, 2,2'-oxybis(1-chloropropane), benzene, and 1,4-dichlorobenzene. Cancer risks for the construction worker were based on a much shorter exposure duration compared to the industrial worker; therefore, did not exceed 1E-4.

For the child resident (HI=43) and adult resident (HI=31), the maximum of the estimated target organ HIs exceeded 1.0, but not for construction and industrial workers. For the residential receptors, the COCs exhibiting HQs greater than 1.0 include 2,3,7,8-TCDD TEQs, Aroclor-1016, aluminum, antimony, arsenic, cadmium, cobalt, copper, iron, manganese, mercury, vanadium, and thallium. Additional COCs contributed to target organ-specific HIs that exceeded 1.0, but individually were associated with HQ contributions of less than 1.0, including zinc, 2,6-dinitrotoluene, 2,4-dinitrotoluene, nitrobenzene, and barium.

Blood lead concentrations were predicted to exceed 10 µg/dL in 100 percent of an exposed population of child residents. Blood lead predictions were not able to be generated for construction workers because the Adult Lead Model is not calibrated for groundwater exposure.

#### 6.1.5 Uncertainty Analysis

There are various uncertainties associated with any risk assessment. Most uncertainties identified for the HHRA will result in the potential for overestimation of risk for both the RME and CTE scenarios. Further information regarding site-specific risk assessment uncertainties is discussed in Appendix U, Part 1,



Section 7.8. The following uncertainties should be considered as part of any risk management decisions for the site:

- Impacts of Background Levels: Background tests were performed after site risks were presented for soil COPCs (see Appendix U, Part 1). RAGS D Table 10s display all of the risk drivers that contribute to the significant cancer and noncancer risks, with footnotes added to inorganic substances found to be within the range of background based on statistical analysis. Organic compounds that were similar to background were not flagged on Table 10s; however, anthropogenic contributions from the urban environment are not expected to represent the bulk of detected soil concentrations for any substance. Groundwater data from the Clearview Landfill could not be compared to upgradient groundwater conditions because Clearview Landfill is situated at a topographic high point relative to surrounding areas, and creeks and streams surrounding the landfill are at lower elevations. Therefore, non-site related influences to groundwater, whether naturally occurring or anthropogenic, cannot be readily determined. The background urban environment surrounding the landfill may have contributed to concentrations detected in Darby Creek, Cobbs Creek, and Tinicum Marsh. However, the HHRA did not attempt to characterize background levels of COPCs in surface water and sediment since urban background contaminant patterns are likely to be non-uniform, which would have led to a poorly representative data set, and could bias statistical comparisons of the site versus background for surface water and sediment.
- Uncertainties Regarding the Estimation of the EPC: Several issues can introduce inaccuracies in the calculation of the EPCs for an area of interest. The EPA software program ProUCL, version 3 was utilized to estimate 95% UCLs. This version of the software does not impute non-detected values in a statistical manner, but instead relies on an across the board approximation which assumes one-half the detection limit concentration is present for any sample reported as non-detected, which tends to degrade accuracy as the proportion of non-detects in the data set increases. Another uncertainty related to EPCs is associated with UCL calculations based on data sets having very few (such as less than 8 to 10) detected sample results or data sets with too few sample results to allow any statistical calculation of a UCL. This occurred for a few organic COPCs in groundwater and soil, although nearly all of the major risk drivers were associated with an acceptable number of detected results used to calculate the UCL.
- Uncertainties in Laboratory Data Quality: Validated laboratory data were used to calculate EPCs for groundwater, soil, surface water, and sediment. Therefore, rejected and blank-qualified data were eliminated from the risk assessment to avoid use of unreliable results. However, in a few cases, such as with methylene chloride and phthalates, this step reduced the total number of usable data points available to characterize an area of interest.
- Uncertainties in Risk-Based Screening Levels: RBCs from October 2007 were used to select COPCs, but these criteria have been revised by EPA as recently as December 2009. In certain cases, changes to RBCs could result in elimination of a candidate COPC or inclusion of a new COPC. However, in the HHRA, the estimated risks were based on the latest noncancer RfDs and cancer SFs; therefore, the inclusion of too many COPCs would not cause inaccuracy in the final conclusions of the risk assessment. This uncertainty is evaluated further in Appendix U, Part 1, Section 7.5.
- Uncertainties Associated with Exposure Assessment: The likelihood of the occurrence of the defined exposure scenarios is a source of uncertainty. The future anticipated land use near the landfill is expected to remain as residential in Zone 3 (existing Eastwick Neighborhood), recreational in Zone 2 (existing City Park), or industrial and possibly recreational in the case of

Zone 1 (Clearview Landfill). All of these receptors, plus the construction worker, were evaluated to provide a comprehensive assessment of potential current and future risks from exposure. In addition, the exposure assessment includes various models and equations used to estimate exposure doses or contaminant concentrations, and include several physical parameters that cannot be measured precisely. For example, there is uncertainty in the use of modeled air concentrations (e.g., estimated indoor air and outdoor air breathing zone concentrations as a result of volatile emissions from soil gas or groundwater) in place of monitored values, which may not be indicative of actual site conditions during exposure.

- Uncertainties Associated With Toxicity Assessment: Uncertainty is associated with the RfDs and SFs because of the extrapolation of animal data to humans, the extrapolation of carcinogenic effects from the laboratory high-dose to the environmental low-dose scenarios, and interspecies and intraspecies variations in toxicological endpoints caused by chemical exposure. The use of EPA RfD values is generally considered to be conservative because the doses are based on no-effect or lowest-observed-effect levels and then further reduced with uncertainty factors to increase the margin of safety by a factor in the neighborhood of 10 to 1,000 fold. In most cases, toxicity assessment uncertainties tend to generate over-estimated risks rather than underestimates.
- Uncertainties Associated With Lifetime Recreational Exposure: While the long term exposure duration for all potential receptors is highly variable, a conservative approach was to assume that recreational receptors may visit the site over a 30 year period, including 6 years as a child and 24 years as an adult. Unless the recreational user also happened to be a nearby resident, it would be rather unlikely that the majority of recreational receptors would frequent the site over this long of a timeframe. Therefore, lifetime recreational receptor cancer risks may tend to be overestimated.
- Uncertainties Associated With Blood Lead Modeling: The child and adult blood lead models applied to this risk assessment were developed and calibrated for soil and groundwater exposure modeling. Risks could not be estimated for lead exposures in surface water and sediment. In the groundwater data set for lead, the mean lead concentration was 1,150 µg/L and 14 out of 200 sample results exceeded 1,000 µg/L. These levels are well above the levels encountered in typical groundwater samples. As a result, the IEUBK model reported uncertainty in the accuracy of predicted blood lead levels because environmental exposures associated with blood lead levels above 30 µg/dl are above the range of values used in the IEUBK model's calibration and verification.

## 6.2 Ecological Risk Assessment

Ecological risk assessments (ERAs) were conducted to determine potential impacts of site-related contaminants to ecological receptors. The ERAs were conducted in accordance with the concepts, approaches, and methodologies described in the *Ecological Risk Assessment Guidance for Superfund (ERAGS): Process for Designing and Conducting Ecological Risk Assessments* (EPA, 1997). The ERA process consists of eight steps; a) Steps 1 and 2, which constitute a Screening Level ERA (SLERA), followed by four additional steps (Steps 3 through 7), which constitute a Baseline ERA (BERA), and conclude with Step 8, a risk management evaluation. The first step of the BERA (referred to as Step 3 herein) includes a refinement of the conservative exposure assumptions and is often included as part of the SLERA. Steps 3 through 7 are conducted if additional evaluations or investigations are necessary based on the results of the SLERA. Aspects of Step 8, risk management evaluation, are addressed throughout the ERA process.

The SLERA was conducted to evaluate potential hazard from chemical concentrations detected in media (i.e., soils, surface water, sediment, etc.) associated with the Clearview Landfill area (Tetra Tech/Black & Veatch, 2006). Because potentially unacceptable risks to ecological receptors were identified in the SLERA, a BERA was conducted (Lockheed Martin/REAC, 2008). The following sections summarize the findings from the SLERA and the BERA.

The conceptual site model depicted in Figure 6-5 was developed for the source of contamination, contaminant release and transport mechanisms, affected media, known and potential routes of migration, and ecological receptors. In addition, the site-specific potential pathways for surface soil and groundwater, and surface water and sediment are shown in Figures 6-6 and 6-7, respectively.

Note that, for this RI report, the term “contaminants of potential concern (COPCs)” was used to be consistent with the term used in the SLERA and BERA. However, the final COPCs from the BERA are typically referred to as contaminants of concern (COCs) and evaluated further in the FS.

#### 6.2.1 Screening Level Ecological Risk Assessment (SLERA)

The SLERA was conducted to evaluate potential risks from contaminants associated with the Clearview Landfill to surrounding ecological receptors. The ecological setting is fully described in Section 2.1 of the SLERA report (Appendix V). The study area consisted of various terrestrial and aquatic habitats, including the John Heinz NWR, which contains the largest remaining freshwater tidal marsh in Pennsylvania (see Figure 3-13). This section summarizes general approach employed and the findings from the SLERA.

##### 6.2.1.1 General Approach

This section presents the general approach that was followed as part of the SLERA. The SLERA evaluated chemical data in surface water, groundwater, sediment, and surface soil. The samples were collected in the following habitats, as shown in Figure 3-13:

- Maintained Lawn/Open Field (City Park)
- Disturbed Shrub-Scrub/Wooded Area (Clearview Landfill/southern part of City Park)
- Tidal Marsh/Open Water
- Non-Tidal Marsh/Open Water
- Tidal Riverine
- Landfill Seep Areas
- Groundwater

These various habitats were evaluated in the SLERA using the following Assessment Endpoints (AEs):

##### **Terrestrial - Surface Soils**

- Maintained Lawn/Open Field and Disturbed Shrub-Scrub/Wooded Area
- AE No. 1 – Protection of groups of terrestrial plants and invertebrates from the toxic effects (on survival, growth, and reproduction) of contaminants present in surface soils.
- AE No. 2 – Protection of groups of invertebrate-eating animals from the toxic effects (on survival, growth, and reproduction) of contaminants present in prey items.

#### **Aquatic - Sediment**

- Tidal Marsh/Open Water, Non-Tidal Marsh/Open Water, Tidal Riverine, Landfill Seep Areas
- AE No. 3 - Protection of benthic invertebrates and aquatic plants from the toxic effects (on survival, growth, and reproduction) of contaminants present in sediments.
- AE No. 4 - Protection of aquatic insectivore communities from the toxic effects (on survival, growth, and reproduction) of contaminant present in prey items.

#### **Aquatic - Surface Water**

- Tidal Marsh/Open Water, Non-Tidal Marsh/Open Water, Tidal Riverine, and Landfill Seep Areas
- AE No. 5 - Protection of fish and other aquatic organisms from the toxic effects (on survival, growth, and reproduction) of contaminants present in surface water.
- AE No. 6 - Protection of piscivorous communities from the toxic effects (on survival, growth, and reproduction) of contaminants present in prey items.

#### Initial Screening (Steps 1 and 2)

Steps 1 and 2 of the ERA process examined the potential risk of both upper- and lower-trophic receptor guilds. The lower-trophic level receptors, including terrestrial plants and invertebrates, aquatic plants and benthic invertebrates, and fish, were analyzed conservatively by comparing the maximum concentrations detected in site media to literature-based ecological screening values. The upper-trophic level receptors examined included terrestrial birds and mammals, as well as aquatic birds, mammals, and reptiles. Bioaccumulation of COPCs by upper-trophic level organisms and the food they ingest were examined by a food-web model. The use of maximum site concentrations and other conservative exposure parameters were used in the food-web models to conservatively evaluate risk from COPCs in each medium.

COPCs were identified in Steps 1 and 2, based on the exceedances of ecological screening values by maximum detected concentration or ½ the reporting limit used for non-detected contaminants. Contaminants which did not have a screening toxicity value were also retained as COPCs. These retained COPCs were subsequently analyzed in Step 3.

The SLERA also identified complete exposure pathways in which plants and animals are exposed to contamination and a potential for risks, including:

- Direct exposure to preliminary COPCs in surface soil
- Food chain exposure to preliminary COPCs in surface soil
- Direct exposure to preliminary COPCs in sediment
- Food chain exposure to preliminary COPCs in sediment
- Direct exposure to preliminary COPCs in surface water
- Food chain exposure to one preliminary COPC (i.e., cadmium) in tidal riverine surface water

#### Refinement of Exposure Assumptions (Step 3)

The preliminary COPCs were further examined as part of the SLERA (Step 3 of the ERA process), with respect to the complete exposure pathways identified in Step 2. Food chain exposure to COPCs in other surface waters besides the tidal riverine resulted in no potential risk to upper-trophic level organisms, thus this pathway was not further examined.

During this evaluation, the list of preliminary direct and food chain COPCs established during Steps 1 and 2 were further evaluated in order to identify those chemicals that are risk drivers under more typical exposure scenarios. Initial screening values were not available for several chemicals, thus these chemicals were identified as preliminary COPCs. In Step 3, these COPCs were compared to alternate screening values to refine the list of COPCs for direct exposure. Additional criteria were also applied to help refine the direct exposure COPCs, including revised frequency of detection (e.g., how many times a chemical was present out of the total number of samples), revised magnitude of detection (e.g., how high were the actual concentrations of each chemical), and revised details from each animal's dietary habits. Each of these variables that can be revised is outlined in EPA guidance documents (EPA, 1997 and 2001).

For food chain exposure, several conservative assumptions (from the ERA) were made to present a more typical exposure scenario. These assumptions included:

- Use of average concentrations of the samples,
- Use of a smaller area use factor for each receptor (e.g., the site does not represent 100% of the receptor home range),
- Use of refined dietary assumptions (e.g., average body weight and average food ingestion rate).

#### 6.2.1.2 Summary of the SLERA

Based on the findings of the SLERA, it was concluded that risk might exist to lower- and upper-level organisms in the terrestrial, tidal marsh/open water, and non-tidal marsh/open water habitat areas. However, it appeared that there was no risk to fish-eating animals from contaminants in tidal riverine surface water through food chain exposure.

The following presents a brief discussion of the risks identified in each habitat during the SLERA.

##### *Terrestrial Habitat*

The SLERA indicated that there were likely to be risks to lower-and upper-trophic level organisms in the terrestrial areas of the Clearview Landfill area, including Clearview Landfill, the City Park, and the Eastwick neighborhood. The SLERA identified the location of the maximum detected concentrations of the COPCs identified in the terrestrial areas.

Thirty-six contaminants, consisting of 14 metals, 3 PCBs, 2 pesticides, 1 SVOC, and 16 PAHs, were identified within the maintained lawn/open field area of City Park as posing a risk to either lower-trophic level organisms through direct contact or to upper-trophic level organisms through the food-web, and were retained as COPCs after Step 3a. Forty-four contaminants, consisting of 14 metals, 2 PCBs, 10 pesticides, 2 SVOC, and 16 PAHs, were identified within the disturbed shrub-scrub/wooded area of Clearview Landfill as posing risk to either lower-trophic level organisms through direct contact or to upper-trophic level organisms through the food-web, and were retained as COPCs after Step 3a.

##### *Tidal Marsh/Open Water Habitat*

The SLERA indicated that risks are likely to exist to lower- and upper-trophic level organisms in the tidal marsh/open water habitat of the LDCA site, which roughly forms much of the eastern end of Tinicum Marsh at John Heinz NWR. The SLERA also identified the locations of the maximum detected concentrations of the COPCs identified in the tidal marsh/open water area. The SLERA found that 39 COPCs were present within the tidal marsh/open water sediments. Of these, 14 metals, 4 pesticides, 4 SVOCs, 14 PAHs, and 3 VOCs were retained as COPCs. Aluminum and nickel detected within the surface waters of the tidal marsh/open water area also were retained as COPCs.

The available data were adequate to conduct the initial steps of the ERA process for the stream channel of Darby Creek in Tinicum Marsh. Only three samples of sediment and surface water were collected during the RI, and used to evaluate the risks from surface water and sediment within the remainder of the 145-acre tidal marsh outside the channel of Darby Creek. Additional investigations of this area are being conducted as part of the LDCA site-wide evaluation.

The sediment transport evaluations presented in the previous section suggested that at least six times as much water/sediment is transported downstream with normal ebb-tidal flow vs. upstream on the flood-tide. This suggests that the majority of any contaminated sediment derived from Clearview Landfill is present in the Darby Creek channel. However, the movement of contamination with fine-grained sediment or with surface water could occur during flood tide cycles, and thus move contamination into the network of channels in Tinicum Marsh.

#### *Non-Tidal Marsh/Open Water Habitat (Impoundment)*

All of the surface water and sediment data used for the impoundment risk characterization were collected as part of the RI. The available surface water and sediment samples from this area were adequate to begin the ERA process.

The SLERA indicated that risks are likely to exist for ecological receptors in the impoundment from the sediments, but generally not from the surface water. The impoundment is a non-tidal marsh/open water area located in the John Heinz NWR at Tinicum. The SLERA identified the location of the maximum detected concentrations of COPCs in the impoundment. The sediment samples from the impoundment contained 10 metals, 5 pesticides, 4 SVOCs, 11 PAHs, and 2 VOCs that were retained as COPCs. The maximum detections for all pesticides, PAHs, VOCs, and three of the 10 metals that were retained as COPCs were detected at SD32 (see Figure 3-5 for this location).

The SLERA determined that risks are likely to be minimal to ecological receptors from surface water exposures in the impoundment. In surface water, only four metals (aluminum, iron, nickel, and vanadium) exceeded ESVs, based upon detected concentrations for direct exposure. Maximum concentrations for the four metals retained as COPCs were detected at two locations SW32 (aluminum and vanadium) and SW36 (iron and nickel). No COPCs were retained for indirect exposure through the food-web in surface water from the impoundment.

#### *Tidal Riverine Habitat*

This area consists of the Darby and Cobbs Creeks adjacent to and downstream of Clearview Landfill. In the sediment of the tidal riverine area, 9 metals, 1 PCB, 6 pesticides, 8 SVOCs 16 PAHs, and 2 VOCs exceeded their ESVs, or are likely to bioaccumulate higher than “effect level” concentrations, and were therefore identified as COPCs. The SLERA determined that risks from surface water exposures in the tidal riverine area were minimal. In surface water, only aluminum, iron, lead, and nickel exceeded ESVs for direct exposure; therefore, no COPCs were retained for additional evaluation of potential indirect exposures through the food-web. The SLERA identified the location of the maximum detected concentrations of the COPCs identified in the tidal riverine area. No chemicals in the tidal riverine surface water were retained as COPCs for piscivorous wildlife.

### *Landfill Seep Habitat*

This area includes Clearview Landfill forming the east bank of the Darby and Cobbs Creeks. Analysis of the landfill seep area included sediment and surface water sampled at leachate seeps, as well as groundwater closest in proximity to the landfill seep area. The SLERA identified the location of the maximum detected concentrations of the COPCs identified in the landfill seep area. Thirty-four COPCs were subsequently identified in the sediment of the landfill seep habitat, including 8 metals, 7 pesticides, 14 PAHs, 1 SVOC and 4 VOCs. Surface water analysis within the landfill seep area resulted in 13 metals, beta BHC, and cyanide being identified as COPCs, while 10 metals, Aroclor-1260, cyanide, and hexavalent chromium were retained as COPCs within groundwater.

Leachate sediment may present risks to ecological receptors. Exposure of lower-trophic level organisms to leachate sediment is the only complete exposure pathway within the landfill seep area.

In contrast, aqueous leachate does not appear to present a significant risk although many contaminants were identified as COPCs within the leachate from the landfill seep habitat. This occurs because the assessment of potential risks to organisms was based on the assumption of direct exposure to site contaminants. Direct animal/plant exposure to these media is limited due to immediate dilution of leachate seepage with surface waters in creeks; therefore, it was assumed that this is not a 'complete' exposure pathway for ecological receptors.

#### 6.2.2 Baseline Ecological Risk Assessment

Since the SLERA indicated that there was potential for risk to ecological receptors from exposure to environmental media at OU-1, the BERA was conducted to further evaluate COPCs identified during SLERA, focusing on those chemicals that are risk drivers under "typical" exposure scenarios. The following sections describe general approach employed and summarize the findings from the BERA.

##### 6.2.2.1 General Approach

The COPCs retained from the SLERA were further evaluated to determine whether any of the COPCs presented an acceptable risk when other information pertinent to the site is considered. Criteria used to select final COPCs (risk drivers) during the BERA included:

- COPCs that were not detected at the sample quantification limits (SQLs) were eliminated from further evaluation in the BERA.
- Environmental conditions at the site were evaluated to determine whether COPCs identified in the SLERA could be eliminated from further consideration. For example, certain COPCs may only be bioavailable or exist in toxic forms under certain environmental conditions.
- COPCs were considered for elimination when on-site concentrations were not statistically significant or different from concentrations at reference stations (Darby and Cobbs Creeks upstream of Clearview Landfill).
- COPCs were considered for elimination when on-site concentrations were not statistically significant or different from published background or baseline concentrations.
- COPCs with hazard quotients (HQs) greater than 1.0 were retained as final COPCs. Note that alternative benchmarks (if available) were used to screen COPCs for which EPA Region III benchmarks were not available.
- Any COPCs with no toxicity information were retained and discussed as an uncertainty

The AEs and corresponding measurement endpoints evaluated in the BERA are described below:

- **AE No. 1 - Survival and Growth of Aquatic Plants**  
Measurement Endpoint: measured COPC concentrations in sediments along with associated physical/chemical measurements were compared to toxicological benchmarks for terrestrial plants (Efroymson, *et.al.* 1997a).
- **AE No. 2 - Survival, Growth, and Reproduction of Terrestrial Plants**  
Measurement Endpoint: measured COPC concentrations in soils along with associated physical/chemical measurements were compared with toxicological benchmarks for terrestrial plants.
- **AE No. 3 - Survival, Growth, and Reproduction of Aquatic Invertebrates**  
Measurement Endpoints: (a) a 42-day chronic sediment toxicity tests were conducted using the amphipod, *Hyallela azteca*. Effects of exposure to site sediments and control sediments on the survival, growth, and reproduction of the amphipod were compared; (b) a life-cycle test was conducted using the midge, *Chironomus tentans*. The effects of exposure to site sediments and control sediments on the chironomid were compared; and (c) a 28-day *Lumbriculus variegatus* bioaccumulation test was conducted with sediments collected from Darby Creek adjacent to the Clearview Landfill, to determine whether the COPCs in the sediments accumulate in aquatic invertebrates.
- **AE No. 4 - Survival, Growth, and Reproduction of Terrestrial Invertebrates**  
Measurement Endpoints: (a) measured COPC concentrations in soils were compared with toxicological benchmarks for soil microorganisms, earthworms, and other soil-dwelling organisms; (b) a 28-day earthworm (*Eisenia fetida*) bioassay was conducted. Survival in site soils and control soils were compared; and (c) an earthworm (*Eisenia fetida*) 28-day bioaccumulation test was conducted to determine whether the COPCs in the soils at the Site accumulate in terrestrial invertebrates.
- **AE No.5 - Survival, Growth, and Reproduction of Fish**  
Measurement Endpoints: (a) measured COPC concentrations in surface water along with associated physical/chemical measurements were compared with surface water toxicological benchmarks; and (b) an acute (96-hour) fish bioassay using the fathead minnow, *Pimephales promelas*, was conducted.
- **AE No. 6 - Survival, Growth, and Reproduction of Aquatic Feeding Birds and Mammals**  
Measurement Endpoint: concentrations of COPCs in the tissues of prey species (*i.e.*, whole body tissue residues based on *Lumbriculus* sp. bioaccumulation site data), sediment, and surface water were measured. The risks from dietary exposure to COPCs on-site were determined using dietary exposure models. Exposure doses were calculated using dietary models for sediment-probing birds (Lesser scaup) and aquatic feeding mammals (raccoon), incorporating COPC concentrations in prey, sediment and surface water, compared with TRVs derived from the literature.
- **AE No. 7 - Survival, Growth, and Reproduction of Terrestrial Feeding Birds and Mammals**  
Measurement Endpoint: concentrations of COPCs in the tissues of prey species (*i.e.*, tissue residues based on earthworm bioaccumulation site data), site soil, and surface water were measured. The risks from dietary exposure to COPCs on-site were determined using dietary exposure models. Exposure doses were calculated using the dietary models for soil-probing birds



(American robin) and terrestrial feeding mammals (short-tailed shrew), incorporating COPC concentrations in prey, sediment and surface water, were compared with TRVs derived from the literature.

#### 6.2.2.2 Results of the Site-Specific Toxicity/Bioaccumulation Tests

This section summarizes the results of the toxicity and bioaccumulation tests that were conducted as part of the BERA. More detailed information is available in the BERA report (Appendix V).

##### Soil Tests

**42-Day Potworm Toxicity Test.** Seven soil samples from the City Park and background area (see Figure 3-13 for sampling locations) were collected and submitted to Aqua Survey Incorporated for the toxicity tests using *Enchytraeus albidus* (enchytraeid Potworm). The tests were performed in accordance with the American Society of Testing and Materials (ASTM), method E1676-04. Conclusions from the Potworm tests are summarized as follows:

- Of the seven soil samples tested, only two resulted in significantly reduced Potworm survival after 21 days. The soil samples from GP081 and background resulted in 58% and 60% survival, respectively.
- The results for the 42-day reproduction endpoint indicated three site samples (from GP-031, GP-081, and GP-032) and the background sample had significantly lower reproduction compared to the control sample,

Note that the BERA did not evaluate the potworm test results. However, these results were further evaluated and deemed appropriate to characterize risks to soil invertebrates. Therefore, the potworm test results will be considered during the remedial planning process identifying ecological risk drivers and their cleanup goals.

**28-Day Earthworm Toxicity Study.** Seven soil samples from the City Park area were collected and submitted to Great Lakes Environmental Center (GLEC). Earthworms were grown in these soil samples to perform a long-term toxicity exposure study in accordance with ASTM method E1676-04. At the end of the test, the earthworm samples were submitted to CompuChem (LIBRTY) for chemical analysis of inorganics and benzaldehyde.

- The results of this study indicate no effects on earthworm mortality after 14 days, as all samples resulted in 100% survival. No long-term effects on mortality were observed after 28 days because all samples resulting in 28-day survival rates of >90%, with the exception of GP032. Sample GP032 resulted in a 28-day survival rate of 81.7%, which may indicate a long-term effect on the survival of terrestrial invertebrates associated with soil in this sampling location. The difference in weight between test initiation and termination was similar for the samples from all locations when compared with laboratory control samples, indicating no long-term effects on growth. Reproduction was not a measured endpoint in this study.
- Table 6-2 summarizes the concentrations of the chemicals detected in the tissue of worms used in 28-day bioaccumulation testing, as well as the tissue concentration of the initial, control, and background worms. As indicated in Table 6-2, some contaminants were detected in site worms at only a small increase (<25%) or at concentrations less than that detected in the worms used in the

test control samples. Only one SVOC, benzaldehyde, and all (20) metals were detected in at least one tissue sample. The tissue sample from GP031 had the highest measured concentration of eight metals including aluminum, barium, chromium, cobalt, iron, manganese, nickel, and vanadium.

**Field-Captured Invertebrate Tissue Analysis.** Five invertebrate tissue samples were collected from GP021, 031, 032, 081, and background location, and submitted to CompuChem laboratory for tissue analysis of inorganics, SVOCs, and pesticide/PCBs. Inorganics, SVOCs, and pesticides/PCBs were analyzed under the Contract Laboratory Program (CLP). Table 6-3 summarizes substances detected in tissue of earthworms collected at the sampling locations.

- Earthworm tissue from the sampling location GP031 near the Angelo Place townhouses contained the highest concentrations of 12 of the 14 detected SVOCs detected among the tissue samples collected for ecological evaluations.
- The maximum concentrations of benzo(b)fluoranthene and benzo(k)fluoranthene were detected in the tissue sample from GP081 north of the landfill.
- Earthworm tissue from sampling locations GP021, GP031, and GP032 resulted in the maximum detected concentrations of the 11 detected pesticides, while tissue from GP031 resulted in the maximum detection of Aroclor 1260 which is the only PCB detected. Maximum metals concentrations in earthworm tissue were more variable, but the sample from GP081 had the maximum detected concentrations of 10 of the 22 detected metals in field-caught earthworm tissue.

#### Sediment Tests

**28-Day *Lumbriculus* Bioaccumulation Test.** Seven sediment samples along the Darby/Cobbs Creeks (SD19, SD22, SD24, SD25, SD28, and SD30) and from background location (see Figure 3-13 for sampling locations) were collected and submitted to GLEC. Lab-supplied *Lumbriculus variegatus* creek bottom-dwelling worms were grown in these sediment samples during the EPA 28-day bioaccumulation test method.

- Several effects were measured on these sediment-living worms. Of the seven sediments tested, only one sample resulted in significantly lower survival in the 4-day screening toxicity test: worms in the sediment sample from location SD30 (from Darby Creek near the former Delaware County incinerator). After 28 days, the average wet (depurated) weight of surviving *Lumbriculus* worms was below the required 100 grams needed for tissue analysis for two sediment samples each from SD25 (near the Tank Farm) and SD30. The results of the tissue analysis, summarized in Table 6-4, indicated that many contaminants were present in worm tissue at concentrations only slightly above or below those detected in the tissue at test initiation, except for several metals such as aluminum, barium, and iron.
- Seven pesticides (alpha-chlordane, beta BHC, dieldrin, heptachlor epoxide, DDD, DDE, and DDT), one SVOC (bis[2-ethylhexyl]phthalate), and three metals (arsenic, cadmium, and mercury) were detected in the *Lumbriculus* worm tissues associated with site sediment, yet undetected in the control or initial worm tissues. The tissue of the worm grown in the sediment sample from SD22 (near leachate seepage) had the highest detected concentrations of 19 contaminants, including all metals analyzed, with the exception of arsenic, selenium, silver, and zinc.

**Life-Cycle *Chironomus tentans* Assessment Test.** Seven sediment samples were collected and submitted to Aqua Survey, Inc. (ASI). The samples were utilized to perform the “*Life Cycle Test for Measuring the Effect of Sediment-Associated Contaminant of the Freshwater Midge, Chironomus tentans.*” These lab-supplied freshwater midges were grown in water/sediment from the Darby and Cobbs Creeks, and their life cycles were evaluated.

- No impacts were observed to freshwater midges as a result of exposure to site sediment with respect to survival and emergence. However, two midge samples exposed to the sediment samples each from SD24 and SD25 indicated a reduction in reproduction compared to the controls.
- Of the seven samples tested, only one sample with the background sediment sample resulted in significantly lower survival in comparison to the controls. There was no significant difference in the growth (measured as ash-free dry weight) of larvae from any of the sediments tested in comparison to the laboratory control samples. The emergence (the ability of larvae to become free-flying adults) of organisms was different than the control only for background sediment sample with a significant survival impact. Midges in the two sediments (SD24 and SD25) resulted in significantly lower reproduction compared to those in the controls. No statistical analyses of freshwater midge reproduction were completed for the background sediment sample because it resulted in statistically significant survival differences.

**42-Day Life-Cycle *Hyalella azteca* Assessment Test.** Seven sediment samples from the site were collected and submitted to ASI for the test. The samples were utilized to perform the *42-Day Test Using Hyalella azteca for Measuring the Effects of Sediment Associated Contaminants on Survival, Growth, and Reproduction.* These freshwater amphipods were grown in water/sediment from Darby and Cobbs Creeks, and their life cycles were evaluated.

- Impacts of exposure to site sediment on survival of freshwater amphipods were observed. In the first 28 days of the experiment, organisms in four samples each from SD22, SD24, SD28, and SD30 had significantly lower survival in comparison to organisms in the control sample. Survival observations at days 35 and 42 indicated that survival in only three samples from SD22, SD28, and SD30 continued to be significantly different. Control survival decreased between days 28 and 35 while survival in SD22 did not. Therefore, amphipod survival in the sediment sample from SD-22 at days 35 and 42 was not significantly different than that observed in the controls.
- There was no significant difference in the growth (measured as ash-free dry weight) or reproduction (mean number of young per surviving females) of larvae from any of the sediments tested in comparison to the laboratory control samples. Neither growth nor reproduction impacts could be determined for sediment samples SD22, SD28, and SD30 because the reduced survival results would confound that evaluation.

#### Landfill Seep/Surface Water Tests

**Fathead Minnow Acute Toxicity Study.** Three water samples, including two leachate samples from LS/LC02 and LS/LC05 (where the highest concentrations of COPCs were observed during the SLERA), and one surface water sample from SW22 near seepage, were collected. These samples were submitted to GLEC for the toxicity test in which minnows are grown in site water samples. The samples were utilized to perform 96-hour acute fish bioassay using the fathead minnow (*Pimephales promelas*) according to

Methods for Measuring the Acute Toxicity of Effluents and Receiving Waters to Freshwater and Marine Organisms (5<sup>th</sup> Edition, Oct. 2002).

- No acute toxicity effects were observed for the fathead minnow (*Pimephales promelas*) in the three water samples submitted.

#### 6.2.2.3 Summary of BERA Conclusions

The risk characterization portion of the BERA integrated exposure and effects data, to estimate risks to the assessment endpoints. Risks for each assessment endpoint are summarized below. Note that the reproduction assessment endpoints were not included for some receptors; therefore, there is uncertainty in the risk evaluation for those receptors.

#### **AE No. 1 - Survival and Growth of Aquatic Plants**

Four COPCs (Hg, Se, Tl, and 4-methylphenol) in creek sediment were determined to have HQs exceeding one, based on maximum sediment concentrations. The HQs of the same four COPCs also exceeded one for the reference area. There were seven COPCs with HQs exceeding one in Tinicum Marsh, while there were four COPCs with HQs exceeding one in the Impoundment area. Four COPCs (endosulfan sulfate, bis[2-ethylhexyl] phthalate, dibenzofuran, and acetone) were not characterized because benchmarks for these COPCs were not available. However, the maximum concentrations and the frequency of analytical detections for these four COPCs were all similar to those in the reference area (Darby and Cobbs Creeks upstream of the Clearview Landfill).

#### **AE No. 2 - Survival, Growth, and Reproduction of Terrestrial Plants**

Soil risk to the terrestrial plants was characterized at the Landfill-Business Area (a.k.a., southern industrial area), Landfill-Wooded and Shrub Area (Clearview Landfill), and City Park containing many COPCs with HQs exceeding one. A total of 27 COPCs for the Landfill-Business Area, 29 COPCs for the Landfill-Wooded and Shrub Area, and 24 COPCs for City Park had HQs exceeding one, based on the maximum soil concentrations. Because no site-specific biological studies were conducted for plants, this assessment endpoint was only evaluated by comparing chemical concentrations to conservative screening levels.

#### **AE No. 3 - Survival, Growth, and Reproduction of Aquatic Invertebrates**

Adverse effects of site sediment to aquatic invertebrates in the *H. azteca* were evident from a decline in survival of *H. azteca* for the sediment samples collected at SD22, SD28, and SD30 with 42-day survival of 62.5%, 62.5% and 81.3%, respectively. Neither growth nor reproduction impacts could be determined for these three sediment samples because the reduced survival results would confound that evaluation.

Adverse effects to aquatic invertebrates in the *C. tentans* study was limited to a slight depression of reproduction (total number of eggs produced per female) for *C. tentans* in two sediment samples (SD-24 and SD- 25) compared with the control group reproduction. In summary, risk to the aquatic invertebrates exposed to COPCs in creek sediment exists, based on the toxicity studies, particularly the reduced survival of *H. azteca* at three of the six sampling locations in Darby Creek.

#### **AE No. 4 - Survival, Growth, and Reproduction of Terrestrial Invertebrates**

Soil risk to the terrestrial invertebrates was characterized at three areas (Landfill-Business Area, Landfill-Wooded and Shrub Area, and City Park). A total of 19 COPCs for the Landfill-Business Area, 22 COPCs for the Landfill-Wooded and Shrub Area, and 12 COPCs for City Park had HQs exceeding one, based on the maximum soil concentrations. The highest HQs for all COPCs occurred within the Landfill-Business Area, with the exception of Pb and Hg in City Park.

In the *E. fetida* toxicity test, there was no significant difference in survival and growth between the laboratory control sample and the site samples, indicating no significant risk of toxicity to terrestrial invertebrates at the site. In addition, although not evaluated in the BERA, soil samples were also collected for a 42-day potworm toxicity test using *Enchytraeus albidus*. The endpoints of this test were reproduction and survival. For the potworm test, one site soil sample (GP081) and the background sample had lower survival after 21 days compared to the laboratory control sample, and three site sample (GP031, GP032, and GP081) and the background sample had lower reproduction compared to the laboratory control sample. Overall soil risk to the terrestrial invertebrates found in the BERA was determined by the number of COPCs that exceeded their benchmark values. The Landfill-Business Area, and Landfill-Wooded and Shrub Area would be considered to pose greater risk to the invertebrates than would City Park.

#### **AE No. 5 - Survival, Growth, and Reproduction of Fish**

Only two chemicals were retained as COPCs in surface water (Cd and Zn) for the BERA. Surface water risk to aquatic life indicated a HQ of 88 for Cd, based on maximum water concentration. In the 96-hour static renewal acute fathead minnow test, survival ranged from 85% to 95% for the fathead minnows exposed to two seep water samples and the surface water sample collected just downstream of the seeps.

For the groundwater samples, all COPCs had HQ exceeding 1.0 with the exception of two analytes (i.e., endrin ketone and benzo[g,h,i]perylene). This assessment was based on compositing all of the groundwater samples collected during all sampling events and utilizing the maximum concentrations for the risk calculations. The movement of these COPCs from the groundwater to Darby Creek is not known; therefore, the potential for risk could not be appropriately characterized.

Risk to the fish community was determined for one COPC (Cd) exceeding the benchmark at maximum concentration in the surface water. Furthermore, the potential movement of COPCs from the groundwater to Darby Creek may pose risk to aquatic life given the high number of COPCs that exceeded the benchmarks for groundwater.

#### **AE No. 6 - Survival, Growth, and Reproduction of Aquatic Feeding Birds and Mammals**

For this assessment endpoint, dietary exposure concentrations were modeled using the Lesser scaup and the raccoon as the receptor species. Three exposure scenarios were modeled as described in the BERA. Model 1 defined the abiotic exposure using the maximum concentrations of COPCs in sediment and surface water using conservative life history parameters. Model 2 calculated total exposure using maximum concentrations of sediment, surface water, and food (tissue), as well as conservative life history parameters. The food intake was based on the *Lumbriculus* bioaccumulation analyses. Model 3 calculated total exposure using maximum concentrations of sediment, surface water, and food (tissue) and representative life history parameters. The modeling was applied to both receptor species exposed to COPCs in Darby Creek adjacent to the landfill, landfill seeps, and the reference area.

When the Lesser scaup was modeled as the surrogate receptor for this assessment endpoint, risk from abiotic exposure (Model 1) to the aquatic feeding birds was determined for dibenzofuran in Darby Creek near the landfill and the reference area. When the dietary component using the site-specific *Lumbriculus* tissue was applied using Model 2, risk to aquatic birds was driven by Hg, Se, and dibenzofuran in Darby Creek and the reference area, but by Hg and Se in landfill seeps. When Model 3 was applied with the more representative life history parameters, model-calculated risk may exist from Hg and Se exposures at all areas, including the reference area. Possible risk may also exist from exposure to Cr in landfill seeps.

When the raccoon was modeled as the receptor for this assessment endpoint, risk from abiotic exposure (Model 1) to the aquatic feeding mammals was determined for dibenzofuran in Darby Creek and the reference area. When the dietary component using the site-specific *Lumbriculus* tissue was applied in Model 2, risk to aquatic mammals was driven by Se and dibenzofuran in Darby Creek and the reference area, but by Se in landfill seeps. Model-calculated risk may exist from Hg, Se, dibenzofuran, and dieldrin in Darby Creek and the upstream reference area, based on Model 2 and/or Model 3. Possible risk may also exist from exposure to Hg, Ni, Se, and dieldrin in landfill seeps.

#### **AE No. 7 - Survival, Growth, and Reproduction of Terrestrial Feeding Birds and Mammals**

For this assessment endpoint, dietary exposure concentrations were modeled using the American robin and the short-tailed shrew as the receptor species. Three exposure scenarios were modeled as described in the BERA. Model 1 defined the abiotic exposure using the maximum concentrations of COPCs in soil and surface water using conservative life history parameters. Model 2 calculated total exposure using maximum concentrations of soil, surface water, and food (tissue) as well as conservative life history parameters. The food intake was based on the earthworm bioaccumulation analyses. The laboratory bioaccumulation test data were used for the metals, while the field-collected invertebrate tissue data were used for the organic chemicals. Model 3 calculated total exposure using maximum concentrations of soil, surface water, and food (tissue) together with representative life history parameters. The modeling was applied to both receptor species at the Landfill-Business Area, Landfill-Wooded and Shrub Area, and City Park.

A greater number of COPCs posed risk or possible risk to the terrestrial mammals as compared with the terrestrial birds. When the American robin was modeled as the receptor for this assessment endpoint, risk from abiotic exposure to the terrestrial feeding birds was determined for Cu and Pb in the Landfill-Business Area, and Pb in City Park. When the dietary component using the site-specific earthworm tissue was applied, risk to terrestrial birds was driven by Pb and Hg at all three areas; Cu at the Landfill-Business Area; and Cr at the Wooded and Shrub Area.

Model-calculated risk may exist from exposure to Cd, Cr and V, based on Models 2 and 3 at all three areas for the terrestrial feeding birds. Possible risk may also exist for Aroclor 1260 at the Landfill-Wooded and Shrub Area, based on Models 2 and 3.

When the shrew was modeled as the receptor for this assessment endpoint, risk from abiotic exposure (Model 1) to the terrestrial feeding mammals was determined for six COPCs at the Landfill-Business Area, four COPCs at the Wooded and Shrub Area, and three COPCs in City Park. When the dietary component using the site-specific earthworm tissue was applied using Model 2, risk to terrestrial mammals was driven by exposures to nine COPCs at the Landfill-Business Area, 10 COPCs at the Wooded and Shrub Area, and seven COPCs in City Park. When Model 3 was applied with the representative life history parameters, risk to the terrestrial mammals was driven by exposures to four COPCs at the Landfill-Business Area, four COPCs at the Wooded and Shrub Area, and two COPCs in City Park.

**6.0 RISK ASSESSMENT..... 1**

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